HYDROLOGIC-BASED ECOLOGICAL RISK ASSESSMENT OF URBAN, AGRICULTURE, AND COAL MINING IMPACTS UPON AQUATIC HABITAT, TOXICITY, AND BIODIVERSITY

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(ABSTRACT)

Urban, agriculture and coal mining land use/cover impacts upon aquatic habitat, toxicity and biodiversity were investigated in Leading Creek, a 388 km^2 watershed in southeastern Ohio. Abandoned strip mine land (ASML) and active deep underground mines were examined along with abandoned near-surface underground mine land (AUML). The work focused on assessment of aquatic toxicity, water quality, and biodiversity through investigation of associated ecological responses for both treated and untreated AMD. Relations were examined among land use/cover, chemistry, and various ecological and toxicological endpoints. Sources of data (scale 1:24000) included Landsat5 imaging from 1988 and 1994, and directly digitized extents of underground mining activities dating to the 19^{th} century, with more recently created strip mines. USEPA and Ohio EPA qualitative habitat scoring protocols were used. Land use/cover thresholds were established using ASML=3%, AUML=2% to 10%, Urban=3% to 5%, and Bare Soil=3%. Biodiversity was assessed using qualitative benthic macroinvertebrate taxon richness and abundance, for total and EPT groups, respectively.

A better understanding of acid mine drainage (AMD) was demonstrated linking land use/cover, coal bed, sediment, and water column chemistry to aquatic ecotoxicity through examination of the origin and fate of sulfate, magnesium, iron, manganese, and zinc. Key findings in risk assessment of Leading Creek indicated that (1) abandoned near-surface underground mine lands (AUML) were associated with >90% of untreated AMD reaching Leading Creek; (2) degradation to aquatic ecology was primarily associated with water quality degradation due to AMD, not with sediment quality degradation; (3) modest habitat destruction, especially sedimentation effects, were observed for ASML>3%, and urbanization>5% in small subsheds; (4) unique chemical signatures differentiated mining techniques instream; and (5) *in situ Corbicula fluminea* growth rates were dependent upon drainage area.

Sporadic signs of agricultural and urban impacts were indicated from acute toxicity with *Ceriodaphnia dubia* and chronic *in situ* toxicity testing with *C. fluminea*. Both the ecotoxicological tests were shown to be reliable indicators of AMD impact from AUML, on watershed and subwatershed scales. AMD was strongly associated with depressed biodiversity, low pH, and elevated zinc. Ecotoxicity monitoring supported interconnections found between sediment and water chemistry, land use/cover, and biodiversity.

DEDICATION

I dedicate this work to my wife Martha and my son Jacob who sacrificed much over the years it took to complete my studies. They were always supportive.

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Chapter 1 - EXECUTIVE SUMMARY

Many eastern United States Appalachian streams have been impacted by coal mining activities. In these landscapes both point and nonpoint sources of pollution play important roles in the decline of aquatic freshwater ecology. Defining risk on watershed scales at some point requires understanding the legacy of abandoned mined lands (AML) and the aftermath of uncontrolled pyretic acid mine drainage (AMD). Active surface and underground mines equally represent critical components of stream health assessment, juxtaposed with concurrent urban and agricultural activities. Owing to AMD's growing importance and impacts to ecology, the U.S. Environmental Protection Agency singled out acid drainage as the number one water quality problem affecting Appalachia. This research focused on assessment of aquatic toxicity, water quality, and biodiversity through investigation of associated ecological responses to both treated and untreated AMD discharged from active and abandoned mines, respectively.

1.1 OVERVIEW

1.1.1 Ecological Risk Assessment of Coal Mining

In Appalachia most AMD is attributed to mining of high sulfur-bearing bituminous coal where oxidation of pyretic sulfides in the coal seam and surrounding rock produces sulfuric acid. According to Skousen (1996), much of the pyrite (FeS_2) in coal and overburden occurs as small crystalline grains intimately mixed with the organic constituents of coal. With lower-sulfur in lignite, subbituminous, and bituminous coals in western states, concerns there are instead dominated by hard rock mining of copper, gold, uranium and other high sulfide bearing ores.

Comparing U.S. States, Montana and Alabama have the highest average pyretic sulfur content (0.6%) for lignite rank coals and Utah the highest average content for sub-bituminous coals. For soft bituminous coals Kansas (3.8%), Missouri and Iowa (2.7%) rank highest followed by Ohio (2.1%). Traveling farther east, sulfur content tends to decrease among bituminous coal deposits. In comparison, the highest heating value anthracite coals more commonly used for home heating have the lowest average pyretic sulfur contents; Pennsylvania (0.25%), Washington (0.15%), and Virginia (0.13%) (USGS, 1998). Based on the national coal quality database, Iowa has shown the overall highest pyretic sulfur contents up to 16% where levels in eastern Ohio's coal belt have been observed as high as 9%. Pyretic sulfur is the major species and the sulfide of greatest concern in coal and associated rocks, and is highly correlated to the maximum potential acidity of these materials (Skousen, 1996).

Risks to ecology from mining are also a function of the natural buffering capacity of host bedrock. Even the presence of small amounts of limestone and dolomite can offer sufficient protection, offsetting pyrite oxidation and reducing stream acidification (USEPA, 1980; Skousen, 1996). Evidence indicates that exposure of sulfides associated with coal and hard rock mining lead to similar conditions in streams and rivers. Major effects include severe sedimentation from land disturbance and ensuing pyretic AMD discharges. These effects can be viewed as the simple result of simultaneous acceleration of natural erosion and oxidation processes that otherwise act upon these landscapes (Leopold et al, 1964). Ecotoxicological studies have shown that mining can be effectively characterized by two signatures: (1) altered water quality causing direct toxicity to aquatic organisms, or (2) an increased suspended solids load causing physical alterations of habitat and indirect impacts to biota (Herricks et al, 1974).

The Appalachian Regional Commission 30 years ago estimated sources of AMD were broken down by underground mining (58%), surface mining (28%), and (14%) mixed mining, the latter also including coal preparation plants. Of these categories, 71%, 12%, and 17%, respectively, represent the amounts of AMD generated by each source (ARC, 1969). Relatively unchanged since, abandoned coal mines still contribute 80% of AMD today (Skousen, 1996). Since 1945, surface mining has for the most part increased production rates, exceeding underground extraction by 1980. Total deep and surface mine coal production in the U.S. rose to 1100 short tons annually by 1996, second only to China. World coal production represents 25% of the world's total energy production today (OSM, 1999), issuing a prognosis that AMD will likely remain an active area of investigation and risk management for generations to come. With ongoing technological improvements that reduce SO_x air pollutant emissions from coal burning, demand for Appalachia's high sulfur coal will not wane.

Assigning risks to aquatic ecology from coal and hard rock mining is feasible, though on watershed scales it is challenging to define and difficult to achieve (Caruccio et al, 1974; Donigian et al, 1995; Richards et al, 1994, 1996; USEPA, 1996, 1998; USDA, 1998). A first limitation exists in differentiating multiple stresses to biota at a fine enough scale, thereby allowing for identification of reduced biodiversity and root causes that presumably derive from activities upstream. Once causes are illuminated, questions lead to whether or not effective strategies exist that can deliver sustainable restoration, ideally defined by quantifiable improvements to ecology (Cairns et al 1996; Daniels et al, 1995; Donigian et al, 1995; Smith et al, 1998; Suter, 1998b; USEPA, 1996; USDA, 1998). Sparse data and often unknown data quality often limit our success. Adding more difficulty, uncertainty analysis is a key step in ecological risk assessment (USEPA, 1998), but may not be well defined. As an example, in watershed settings with complex hydrogeologic processes driving ecosystem cycles (Allan, 1995), standard sensitivity analysis may not be generally applicable (Melching, 1995). Successful land management will depend on our ability to identify, clarify, and simplify access to key relationships between commingled land uses and their ecological effects downstream.

1.1.2 Abandoned Coal Mines

AML and AMD are long-term, large-scale, and expensive social problems (Younger, 1996). Early mining activities in the United States left behind vast extents of unreclaimed or under-reclaimed abandoned mined lands (USDA, 1985). AML surface disturbances included overt changes to the landscape from surface and underground extraction techniques, resulting in shear walls, denuded footprints, mining overburden and spoils, and concentrated residues or mine gob piles left at the surface (Curtis, 1973, 1979; DeWitt, 1988; Dickens et al, 1985, 1989; USEPA, 1976). Disturbed land and bedrock, and residual aggregates remain the primary sources of acid mine drainage. AMD originates from oxidation of several iron sulfide minerals, but is dominated by pyrite (Evangelou, 1995; Nordstrom, 1982). Once disturbed, AMD may discharge as seeps, baseflow, or overland storm flow for decades to millennia depending on hydrogeology.

Along the Iberian Pyrite Belt in Portugal today, AML-AMD related problems still exist related to mining of iron, copper and gold attributed to miners predating the Romans.

It has been estimated that 20,000 km of rivers in the U.S. are impacted by acid mine drainage where 85% to 90% receive run-off from only abandoned surface and deep mines (Skousen, 1996). Skousen also indicated that in West Virginia 85,000 acres were designated as AML in 1977, or 37% of the total state and federal inventory of disturbed land. By the same time, in all of Appalachia, land disturbed by coal mining activities totaled 1.4 million acres. In response, the U.S. Department of Interior, Office of Surface Mining (OSM) reported that a total of $4.6 billion had been collected and deposited into the Abandoned Mine Land Fund from 1977 through 1998. Of these funds $1.25 billion was still considered unappropriated. This tax revenue is based on coal production in that period and will continue to be levied at least through 2004 based on reauthorization by the U.S. Congress in 1992. Between 1977 and 1996, approximately 79,000 acres of damaged lands had been funded or reclaimed through the AML Fund. Early efforts focused on physical hazards like closing shafts and portals, and stabilizing mining waste piles (Skousen, 1996).

Table 1.1, which excludes Indian territories, shows OSM reclamation grants awarded in 1997 and underscores the continuing magnitude and extent of the AML/AMD problem. Appalachian states comprise 62% of the total U.S. concerns in this area. Wyoming and Illinois had the highest percentage of reclamation grant awards in 1997 of all non-Appalachian States with $22.5 million and $9 million in grants, respectively. Only more recently has interest been shown in mitigating AMD damages with an expected increasing trend (NRCS, 1998). Sediment and erosion control, along with protection of human life and property have been the historical focuses of SMCRA since 1978, receiving the greatest prioritization.

Table 1.1 - Appalachian AML Reclamation Profile
1997 SMCRA Grant Obligation Distributions
Office of Surface Mining (OSM, 1999)

State	Grants Total	% All Grants
MD	$2,923,408	1.5%
AL	$4,653,100	2.4%
VA	$7,198,277	3.7%
OH	$10,570,054	5.5%
KY	$19,959,939	10.4%
WVA	$33,649,269	17.5%
PA	$40,003,688	20.8%
Total	$118,957,735	61.9%

In the U.S., over 50,000 acres remain of high priority physical risks associated with highwalls and AML-sedimentation. OSM has inventoried an additional 120,000 acres of AML designated as low priority risks representing AMD and associated environmental degradation. OSM total aereal estimates may be low in some cases due to use of non-standardized inventories that early on focused on inhabited areas with road flooding exacerbated by AML (personal communication with NRCS, 1998). Distributed among 23 U.S. states, 1997 disbursements from the AML Fund totaled $192 million with 62% concentrated in seven Appalachian states, the top four recipient states in order being Pennsylvania, West Virginia, Kentucky, and Ohio. Recently, OSM reclamation efforts have been moving to increase prioritization for ecological and economic hazards attributed to AMD and AML with a redefined interest in water quality.

Another recent development, ecological restoration can now be viewed as a primary objective in some U.S. Army Corp of Engineer projects related to stream improvements designed to mitigate impacts from AML and AMD. Efforts in AML management by OSM in the late 1970’s represented one of the earliest, largest, and more organized attempts in the U.S. to conduct large-scale watershed risk assessment. This led to attempts at AML rehabilitation across several hydrological, geological, and ecological scales. In reaction to growing concerns about AMD, the February 1998 Clean Water Action Plan put forth by the President’s Office called for increased efforts in stream restoration of AMD impacted areas. With its goal to improve surface water quality in areas with abandoned mines, the EPA, the Office of Surface Mining and the Interstate Mining Impact Commission are together developing a more comprehensive approach to watershed restoration for AML. Through its own abandoned mines program, the U.S. Geological Survey (USGS) also maintains special emphasis in research and monitoring related to an ecological assessment and restoration initiative for AML and AMD impacts.

1.1.3 Coal Mining, Metal Toxicity and Commingled Land Uses

With Title 30 of the Code of Federal Regulations, the 1977 Surface Mining Control and Reclamation Act (SMCRA - 30 CFR 25; up through 1987 amendments and the 1992 reauthorization) has since required active management of surface mining footprints and bonding for reclamation. The national NPDES point source permit program of the Clean Water Act in Title 40 remains in place stipulating categorical treatment standards for wastewaters generated from coal extraction and preparation, the latter involving washing of coal (USEPA, 1976a). It is however not clear if categorical effluent limitations are actually protective of instream water quality criteria and standards for protection of aquatic life.

Few comprehensive ecotoxicological risk assessments have been conducted on watershed scales, particularly in environments with active and abandoned mining. Typically lacking in AMD-AML studies addressing water quality issues is adequate quantification of hydrology and hydraulics emphasized in this work. Development of acute and chronic national aquatic water quality criteria (NWQC) for manganese has been initiated by USEPA. More work needs to be done by way of field studies to quantify manganese toxicity in watersheds impacted by active or abandoned coal mining and other sources. Poorly understood key components of AMD, aluminum and manganese are gaining in prominence as critical pollutants adversely impacting aquatic life (Gensemer et al, 1999; Stubblefield et al, 1998). To date, the evolving effort for manganese NWQC development has been led by the State of Colorado in its attempts to delineate state standards for protection of aquatic life impacted by hard rock mining (Stubblefield et al, 1998). Also of interest is assessing impact upon ecology, if any, due to high dissolved solids loads in treated AMD discharged from active underground mining operations. In this study, water column zinc was found to be sufficiently descriptive of AMD risk to aquatic biota.

Addressing other land uses commingled with mining, economic pressures and declining numbers of farms over the last few decades indicate agricultural and forestry best management practices (BMPs) may not be in widespread use. In Meigs County, Ohio for example, hosting 96% of the Leading Creek watershed studied here, 20% of the land was deemed over erodible limits (USDA, 1983). Together with domestic sanitary wastewaters, historical and modern agricultural and mining activities are believed responsible for most chemical and physical stresses expressed in Appalachian streams (OEPA, 1991a, 1991b; USDA, 1983, 1985, 1998). Of these anthropogenic stresses, the individual, relative, and synergistic impacts upon aquatic ecology though are not well understood nor have their effects been sufficiently quantified *in situ* (Allan, 1995; Daniels et al, 1995; Donigian et al, 1995; Karr, 1991; USEPA, 1996, 1998).

1.1.4 Integrating Watershed-Scale Risk Assessments

Today, we now refer to features of the watershed as environmental or ecological "goods and services", presenting a new management paradigm (Cairns *et al*, 1991, 1996; USDA, 1998). In the new paradigm, we can also begin to view the watershed's components with inherent ecological value (Suter, 1998a). In addition to providing recreation (Smith *et al*, 1998), we look for a watershed's ability to filter a specific pollutant for example or to provide water for drinking. Dollar values will eventually be attached to these attributes much the same way we cost out a sedimentation tank or a filtration plant. It is a natural process then to quantify the watershed's ability to provide these goods and services, and to control them. This underscores the importance of developing sound strategies for quantification of ecological goods and services, as well as the risks that jeopardize or diminish these valuable resources. Because of the complexities of AMD, holistic, watershed-scale risk assessment is necessarily a multidisciplinary problem encompassing several areas of science and engineering. Research here attempted to show how multidisciplinary risk assessments can be better organized and integrated. Development of successful frameworks will require identification of computational tools and methods that can screen and relate many types of data including biological, ecological, toxicological, chemical, physical, hydrogeologic, hydrologic, and hydraulic data sets (USEPA, 1996).

To improve understanding of aquatic ecology and how it relates to the landscape, a wide variety of metrics characterizing watershed health must usually be evaluated (Bicknell *et al*, 1993; USEPA, 1996, 1998). This must typically be done within a broad context of conducting multidisciplinary watershed-scale investigations that combine multi-level approaches (Cairns *et al*, 1991, 1996; Donigian *et al*, 1995, Munkittrick *et al*, 1998; Suter, 1998a; USEPA, 1996; 1998). This dissertation concentrated on assessing impacts to aquatic habitat and biodiversity related through measures of land use and land cover, water and sediment quality, and ecotoxicity. Unlike many multidisciplinary watershed-scale ecological risk assessments conducted to date, this work emphasized hydrologic-hydraulic watershed characterization employing standard engineering principles. This was critical to establishing well-defined quantitative descriptions for land use, flow condition, and pollutant loading rates. An overview paradigm of ecological risk assessment for watersheds is presented in Figure 1.1, borrowed from discussions in Chapter 8. Depicting an integrated hydrologic-based multimedia, multidisciplinary approach, Figure 1.1 builds upon an initial assessment framework developed early on in the study of Leading Creek (Gallagher *et al*, 1999), and a triad approach from sediment toxicity assessment (Chapman 1990).

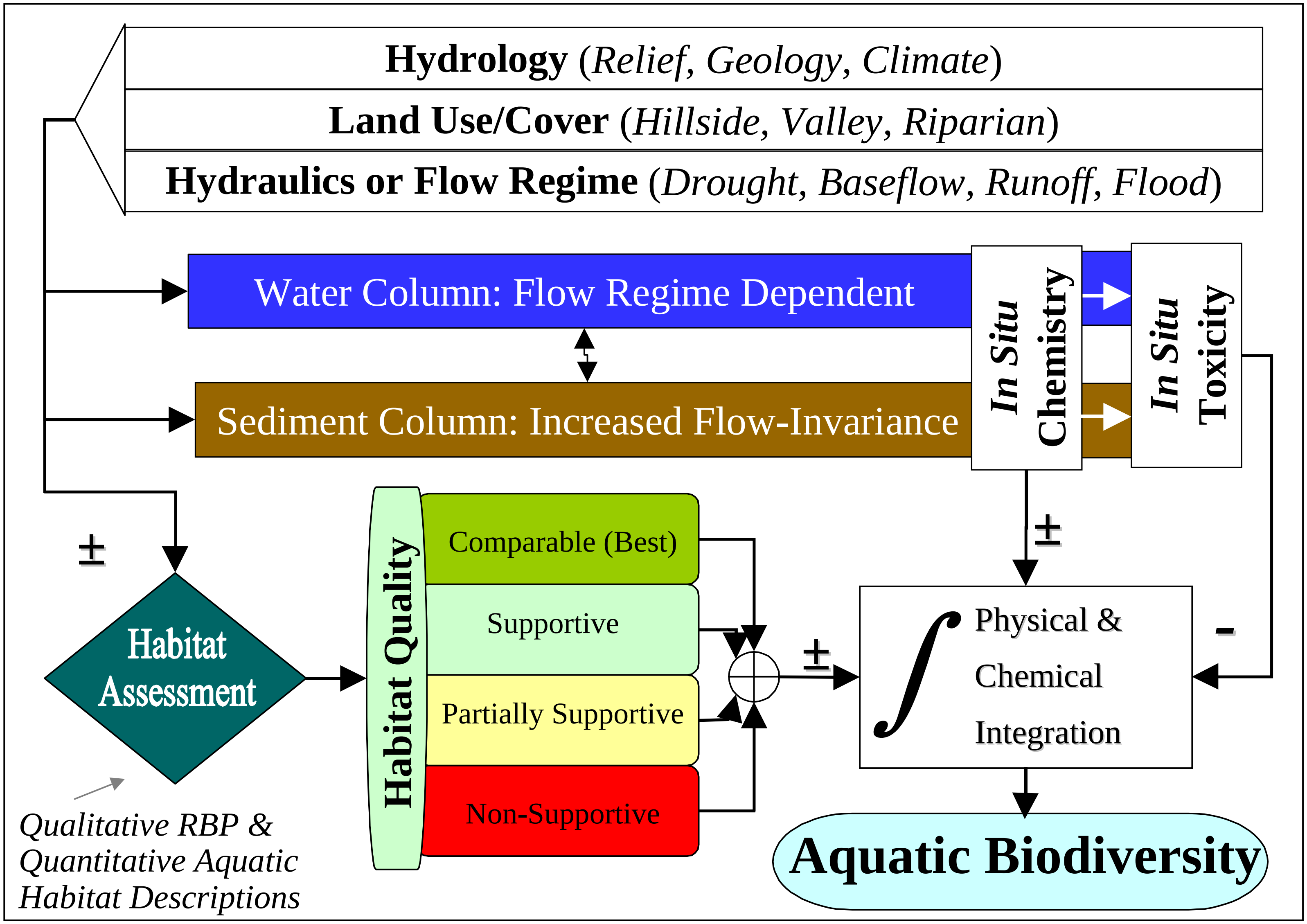

Figure 1.1 – Framework For Hydrologic-Based Watershed Ecological Risk Assessment
- (A Multimedia, Multidisciplinary Hydroecological Approach).

In Figure 1.1, toxicity will always be integrated as a negative impact. Water chemistry can induce both beneficial and negative (non-toxic) effects (e.g. energy and nutrients or turbidity). Habitat can also be positively or negatively influenced by land use/cover and flow regime effects. For example, sedimentation and shifting sands during run-off and flooding can displace aquatic communities. The same flow conditions can cleanse gravel beds of silt and clay, and land uses such as forested areas might offer better riparian quality than agricultural or urbanized landscapes. With respect to its ultimate influence on biodiversity, aquatic habitat will be seen as comparable or supportive (+) in its relationship to biodiversity, or partially supportive to non-supportive (-).

As used here, the concept of risk assessment has been applied in an ecological setting (USEPA, 1992, 1998) as opposed to the more traditional definition of risk analysis associated with evaluating concerns for human health (Vesilind, 1997). Ecological risk assessment has been defined by USEPA as the process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors (USEPA, 1992). Research presented here formulated this process for watersheds from the standpoint of defining anthropogenic stress using variables describing land use/cover characteristics. According to USEPA, ecological risk assessment includes three primary phases: problem formulation, analysis, and risk characterization. Problem formulation includes identifying goals and assessment endpoints, preparing a conceptual model, and developing an analysis plan. Analysis involves evaluating exposure to stressors and the relationship between stressor levels and ecological effect. Risk characterization estimates risk through integration of exposure and stressor-response profiles, describing risk by examining various lines of evidence, determining ecological adversity, and preparing a report (USEPA, 1992). Accordingly, risk assessment evaluates the likelihood of adverse effects while risk management defines a course of action in response to an identified risk.

1.2 HYPOTHESES

Hypotheses investigated here addressed several aspects of hydrologic-based watershed-scale ecological risk assessment of mining environments. The objective was to evaluate quantitative descriptions of agriculture, urban, and surface and underground coal-mining activities, and to relate their impacts to water quality, habitat, toxicity, and biodiversity.

1) At a scale of 1:24000, watershed land-use variables describing urban, agricultural, and various mining activities can be used to characterize associated risks to aquatic ecology.

2) Unique instream water column signatures for agriculture, quarry operations, active underground mining and AML land uses, respectively, can be developed from water quality concentration data, accompanying flow data, and drainage area.

3) *Corbicula fluminea* (Asian clam) growth is known to be dependent upon clam age, stream temperature, and water quality. *In situ* growth rates can also be shown to be spatially dependent where growth rate is expressed as a function of catchment drainage area.

1.3 APPROACH

Broken down into 6 discussion chapters and several supporting appendices, this research developed a comprehensive, hydrologic-hydraulic based ecotoxicological risk assessment of Leading Creek. The work brings together investigations of water quality, environmental engineering, aquatic toxicology, and biology. A formal literature review is first presented offering key background material. Chapters 3 to 8 then present summarized research findings and were developed as manuscripts for publication with intended abridgement. Detailed discussions on hydrologic and hydraulic assessment of the Leading Creek Watershed can be found in Chapter 2 – Literature Review, and the appendices.

The approach used to test hypothesis of this dissertation involved monitoring for a wide range of parameters at 28 to 55 stream sites across the Leading Creek Watershed. Leading Creek, located in southeastern Ohio, is an ungaged 388 km^2 watershed affected by active coal mining, limestone quarry operations, AML sedimentation, AMD, agriculture, and small town urban influences. Three stream sites in two nearby reference watersheds were also evaluated. Research results established an improved capacity to determine the relative risks presented to ecology by commingled agriculture, urban, and assorted mining activities. The work also evaluated GIS applications in environmental and ecological research at finer scales of data collection (scale 1:24000). While variation in scale was not studied in this work, an overall finer resolution for geodata collection was utilized to improve viability of employing a hydrologic-hydraulic based approach. For ecological risk assessment courser scales of investigation (e.g. 1:100000 to 1: 250000 would be expected to be less successful in associating land use/cover variables with metrics of aquatic biodiversity (Richards *et al*, 1994).

Land use/cover data was based on remotely sensed Landsat5 data (1988 and 1994) and directly developed data from paper maps. Drainage area normalized threshold criteria for land use/cover were selected for study. These included AUML (abandoned underground mines) = 2% and 10%, ASML (abandoned strip mines) = 3%, Urbanization = 3% to 5%, Bare Soil = 3%, and Boolean functions based on active mining. Bare Soil was attributed to spring planting and livestock operations. Biodiversity was assessed based on qualitative benthic macroinvertebrate taxon richness and abundance, for total and EPT groups, respectively.

1.4 CONCLUSIONS

A better understanding of AMD was demonstrated linking land use/cover, coal bed, sediment, and water column chemistry to aquatic ecotoxicity through examination of the origin and fate of sulfate, magnesium, iron, manganese, and zinc. Key findings in risk assessment of Leading Creek indicated that (1) abandoned near-surface underground mine lands (AUML) were associated with >90% of untreated AMD reaching Leading Creek; (2) degradation to aquatic ecology was primarily associated with water quality degradation due to AMD, not with sediment quality degradation; (3) modest habitat destruction, especially sedimentation effects, were observed for ASML>3% and urbanization>5% in small subsheds; (4) unique chemical signatures differentiated mining techniques instream; and (5) *in situ C. fluminea* growth rates were dependent upon drainage area.

Sporadic signs of agricultural and urban impacts were indicated from acute toxicity with *Ceriodaphnia dubia* and chronic *in situ* toxicity testing with *C. fluminea*. Both the ecotoxicological tests were shown to be reliable indicators of AMD impact from AUML, on watershed and subwatershed scales. AMD was strongly associated with depressed biodiversity, low pH, and elevated zinc. Ecotoxicity monitoring supported interconnections found between sediment and water chemistry, land use/cover, and biodiversity.

Chapter 3 - Landscape-Habitat: An evaluation was provided to assess relations among land use/cover and aquatic habitat in environments with commingled mining and agricultural activities.

- Two qualitative QHEI habitat subscores, substrate, pool-riffle, and total QHEI were significantly correlated with biodiversity and land use/cover variables studied, where USEPA Rapid Bioassessment Protocol individual and total scores were insignificant.
- ASML>3% and Urban>5% significantly impacted total QHEI scores while active mining and agricultural activities represented insignificant stresses to aquatic habitat.
- For ASML>3% with or without mixed AUML>2%, differences in habitat were attributed to ASML and increased sand fractions, where pool-riffle subscores were also impacted.
- Attributed in part to siltation, for Urban>5% substrate and total QHEI were impacted.

Chapter 4 - Landscape-Biodiversity: Evaluation was next provided to assess land use/cover and biodiversity relationships with initial elucidation of significant differences in water quality.

- AUML>2%, ASML>3%, and Urban>5% showed significant stresses to biodiversity where active mining and agriculture activities represented relatively insignificant stresses.
- Significant differences were observed in biodiversity for AUML>2%, and to a lessor degree for ASML>3%. Results showed total taxa and to a lessor degree EPT taxa were the most sensitive biometrics for both AML land use/cover categories ASML and AUML
- Stress to biodiversity from AUML>2% was attributable to AMD and conditions with low pH. Stress from ASML>3% was attributable to degraded instream habitat and not AMD.
- EPT taxa and EPT abundance biometrics were the most sensitive for Urban>5%.
- Directly developed geodata was more successful than remotely sensed geodata in assessing coal mining impacts to habitat and biodiversity.

Chapter 5 - Coal Bed and Sediment Quality: This work evaluated magnitude of sediment concentrations and ratios of Zn/Fe and Mn/Fe in stream sediments and coal beds.

- Impact upon aquatic ecology attributed to AUML and ASML were shown to be unrelated to sediment quality where concentration levels fell within normal ranges of crustal abundance for %C, Fe, Mn, Cu, and Zn.
- For instream sediments at sites with ASML<3%, LogZn/LogFe ratios were found to be relatively constant across other land uses/covers in Leading Creek as was LogMn/LogFe.
- High LogMn/LogFe ratios and slightly elevated LogZn/LogFe ratios in sediments were tied to ASML>3%, and unusually low LogMn/LogFe was tied to AUML>10% where pH < 6.
- Land use/cover, geologic setting, soil surveys, and chemical quality of mined coal beds were helpful in understanding origin and fate of Fe, Mn, and Zn in stream sediments.

Chapter 6 - AMD Water Column Chemistry: This work evaluated magnitude of water quality concentrations for Fe, Zn. Mn, Mg, TSS, sulfate and various associated product/ratios.

- For a given level of AUML land use (<2%, 2% to 10%, and >10%), quantified gradients in flow regime imparted water quality gradients, elevating AMD pollutants during increasingly dryer conditions associated with baseflow discharges.
- High levels of dissolved Mn can be good predictors of AMD, but not during drought.
- Total water column Zn/Fe can be used to differentiate soluble Zn derived from AMD where Zn and Fe were bound in similar proportion in TSS data as they were in sediments.
- Unique linear relationships between Mg and sulfate were found for limestone quarry operations without AMD impact, for treated AMD without untreated AMD impact, and for untreated AMD without treated AMD impact.
- Zn and sulfate were strongly correlated for untreated AMD associated with AUML.

Chapter 7 - Sediment Ecotoxicity: Ecotoxicological risk among land uses was assessed using whole sediment toxicity tests with *Chironomus tentans* and *Daphnia magna.*

- Sediment toxicity at tributary sites severely impacted by acid mine drainage was not a significant, consistent source of stress to aquatic biodiversity in Leading Creek.
- Sediment toxicity associated with the highest levels of metals in sediment attributed to active mine operations were not a significant source of stress to aquatic biodiversity
- *C. tentans* may be modestly biased (+) by AML impacts attributed to sedimentation.
- *D. magna* may be modestly biased (-) by AUML impacts attributed to AMD and sulfides bound in sediments through *in situ* de-acidification, and later oxidized in test vessels.

Chapter 8 - Water Column Ecotoxicity: Ecotoxicological risk among land uses was assessed using whole effluent acute toxicity tests with *C. dubia* and chronic *in situ* toxicity tests using *C. fluminea* (Asian clam).

- Degradation of biodiversity in Leading Creek induced by toxicants was a function of water quality as opposed to sediment quality.
- Both acute toxicity and *in situ* tests with *C. fluminea* were shown to be reliable indicators of the severity of AMD impact attributable to increased levels of AUML.
- Urbanization in small headwater subsheds and some agricultural influence were also identified with acute water quality toxicity but with more sporadic and inconsistent results.
- The active mining NPDES discharge from Meigs Mine No. 31 associated with well-treated AMD was identified with chronic *in situ* toxicity under lower flow conditions instream. Poor clam response could be attributed to chronic chloride toxicity based on water quality criteria.
- *In situ* shell length growth rates of *C. fluminea* were significantly related to drainage area of the catchment, but not on actual flow rates or relative water velocity from site to site.

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Chapter 2 - LITERATURE REVIEW

2.1 INTRODUCTION

To support the hypotheses investigated in this work, there was a basic need to first describe the fluvial geomorphologic setting and ecological character of eastern United States stream systems like Leading Creek. Discussions below focus on mechanisms that may relate to adverse impacts to the ecosystem associated with agricultural activities (AG) and abandoned mined lands (AML). The survey ends finally with an overview on contemporary techniques available to model and integrate our knowledge of these systems. The literature review covers a wide range of information that may be considered necessary to collectively embody hydrologic-based ecological risk assessment of watershed ecosystems. Research areas that encompass or touch upon the stated hypotheses include:

- Watershed Function and Organization
- Integrated Ecological Risk Assessment
- Watershed Structure and Scale
- Fluvial Geomorphology
- Climate
- Geology, Bedrock, and Soils
- Hydrogeology, Hydrology, and Hydraulics
- Land Use/Cover Analysis
- Historical Impacts of the Ohio River and Leading Creek
- Leading Creek Site Reconnaissance
- Risks to Aquatic Life in Leading Creek
- River Mechanics and Sediment Transport
- Sediment Particle Size in Stream Mechanics and Ecology
- Point Source and Nonpoint Source Pollution
- Agricultural Impacts and Influences
- Agricultural Best Management Practices
- Abandoned Mined Lands
- Acid Mine Drainage
- AML/AMD Reclamation
- Water Quality Assessment
- Sediment Quality Assessment
- Ecotoxicology
- Riparian and Instream Habitat Assessments
- Biodiversity
- Ecological Risk Assessment and Paradigms
- Hydrologic Simulation
- Geographical Information Systems (GIS)
- Expert System Applications for Watersheds
- Water/Sediment/Contaminant Flow/Transport/Fate Modeling
- Analytical and Statistical Methods

2.1.1 Objective-Approach

This study investigated the utility of integrating watershed management, hydrologic analysis, and ecological risk assessment. All three efforts in most standard applications attempt to assess and then manipulate landscape patterns in some manner. This is done often to control an aspect of water quality or quantity, sediment quality or quantity, or to improve ecology (Aronoff, 1993; Bedient *et al*, 1992; Burton *et al*, 1992; Chapman *et al*, 1992; Donigian *et al*, 1995; Dunne *et al*, 1978; Richards *et al*, 1994b; Yang, 1996). Hypotheses were evaluated here to facilitate improved understanding of risk assessment process for watersheds. The Leading Creek stream system was examined in light of two ubiquitous and often dominant land uses, agriculture (AG) and abandoned mined land (AML) (Nordstrom, 1982; OEPA, 1991a, 1991b; USDA, 1983, 1985, 1998; Waters, 1995; Yoder *et al*, 1996). This research showed how biodiversity and bioassessment can be quantitatively related within a hydrologic-based watershed management and risk assessment framework (Allan, 1995; Chapman *et al*, 1992; Suter, 1998a; USEPA, 1992; Waters, 1995).

In this work, measures of hydrology, hydraulics, and environmental engineering were evaluated next to biological data. A primary goal was to describe the ability of engineering tools to compliment and support multidisciplinary ecological-based risk assessment methods. An example of a typical problem faced today is posed by the question "with limited investigative and remedial resources, how do we best improve freshwater aquatic ecology in a stream system spanning perhaps hundreds to thousands of square kilometers?". This study focused on the integration of available engineering and ecological risk assessment methods that try to best answer this question employing watershed and subwatershed investigation scales (Aronoff 1993; McCammon, 1998; Singh, 1995).

2.2 WATERSHED FUNCTION AND ORGANIZATION

We recognize today that almost any action taken in the watershed can enmesh the goals of water resources management, recreational resources management, and sustainable ecology (Cairns *et al*, 1977, 1991, 1996; Doyle, 1997; Dunne *et al*, 1978; NRC, 1992; Shaffer, 1989; Smith *et al*, 1998). According to the U.S. Department of Agriculture, there are more than 3.5 million miles of rivers and streams across the United States. This landscape network of channels combine to form various corridors of economic, social, cultural, and environmental value (Dunne *et al*, 1978; USDA, 1998).

These valued corridors are complex ecosystems that include land, plants, animals, and networks of streams within them. The corridors perform a number of ecological functions such as modulating streamflow, storing water, removing harmful materials from water, and providing habitat for aquatic and terrestrial plants and animals. Stream corridors also have vegetation and soil characteristics distinctly different from surrounding uplands and support higher levels of species diversity (USDA, 1998). Corridors of our rivers and streams equally define hydrologic units of equivalent or greater value. The inherent value of the watershed is ultimately determined by circumstances of its location and the activities that occur within it (Black, 1997; Leopold *et al*, 1964; Linsley *et al*, 1975; Seaber *et al*, 1987). The watershed may also not be independent of activities that occur outside of it, for instance due to impacts from air pollution and acid rain deposition (USEPA, 1980).

Holistic risk assessment must necessarily address socioeconomic issues (Cairns *et al*, 1996; NRC, 1992). Economic trends that may determine watershed character will be defined by industries within and near the watershed. Location theory remains the primary explanation governing why a business or industry will choose one given site over another (Smith *et al*, 1998). Accordingly, location choice factors may include (Shaffer, 1989):

1. A supply or availability of the natural resource that is key for the industry or business use,
2. Other production costs such as quantity and quality of labor available locally or transportation costs for the product,
3. Market proximity,
4. The presence of other supporting businesses that will increase efficiencies, and
5. Personal consideration.

The term "hydroecological" risk assessment is introduced here to underscore the critical and interrelated roles that both hydrology and ecology play in describing watershed function (Bedient *et al*, 1992; Black, 1997; Dunne *et al*, 1978; Linsley *et al*, 1975; Minshall *et al*, 1985; NRC, 1992). Demand for hydroecological engineering assessment techniques, including hydrologic-based ecological modeling, will continue to grow in importance (Donigian *et al*, 1995; Johnson, 1995; Milhous, 1982, 1998; Singh, 1995; Wesche *et al*, 1980). Growing demand for integrated engineering and science watershed applications will add further pressure to develop watershed assessment techniques that can relate landscape activity directly to aquatic ecology (Richards *et al*, 1994b, 1996; USEPA, 1980, 1992, 1998). These techniques will likely continue to evolve and may eventually bring together separate engineering and science approaches currently being used to address these problems (Donigian *et al*, 1995).

2.2.1 Physical, Chemical, and Biological Aspects

Variation in several watershed parameters may compete and combine in the ultimate dominance of biological condition (Karr, 1981a; Karr *et al*, 1985; Minshall *et al*, 1985; Richards *et al*, 1994b, 1996). There are many subtle chemical and physical mechanisms potentially determining biological health (Chapman *et al*, 1992; Karr *et al*, 1981b; Rankin, 1989; Richards *et al*, 1994a, 1996; Rosenberg *et al*, 1986, 1993; USEPA, 1980, 1989; Yoder *et al*, 1996).

Research to date relates the challenge in using chemical and physical data to predict aquatic biodiversity on watershed, subwatershed, and finer scales (Allan, 1995; Bicknell *et al*, 1993; Cherry *et al*, 1995; Milhous, 1982; Richards *et al*, 1994b; Waters *et al*, 1995; Xia, 1997; Xinhao *et al*, 1997). The studies secondly point out that physical, chemical, and biological data may need to be more fully appreciated, in context when defining management schemes to improve ecology (Cairns *et al*, 1991; NRC, 1992; Yoder *et al*, 1996). Physical, chemical, and biological monitoring complement each other (Miller *et al*, 1988) and have determinant contributions to overall ecosystem health (Allan, 1995; Karr *et al*, 1981a, 1981b; Rankin, 1989)

All underlying processes and functions of the entwined physical, chemical, and biological system may need to be investigated to accurately assess damaged ecology (NRC, 1992). Such studies must be conducted on scales that can incorporate critical features of the system, e.g. those that contribute significantly to variation in endpoints (or dependent variables) of interest (USEPA, 1996c, 1998). From the viewpoint of freshwater aquatic ecology, this may often involve typically large watershed-scale studies that can address a wide array of inputs that influence stream condition downstream (Arnold *et al,* 1995; Cairns *et al*, 1996; Donigian *et al*, 1995; NRC, 1992).

2.2.2 Complexity of Watershed Assessments

Ecological risk assessments appear to retain less consistency between studies than one might expect, and in some cases may be quite simple (Chapman, 1995). This may relate to the complexity of the watershed ecosystems under study and the list of variables that can afford to be investigated (Bicknell *et al*, 1993; Chapman, 1990; Cherry *et al*, 1995; Donigian *et al*, 1995; USEPA, 1989, 1992, 1998). The situation illuminates the relatively undefined framework existing today for conducting ecological risk assessments, particularly for complex systems with multiple stressors (Munkittrick *et al*, 1998; Suter, 1998a; USEPA, 1992, 1998). The initial stages are being set for conducting fruitful integrated ecological risk assessments, but there remains a limited ability in current approaches to clearly identify dominant, controlling causes and effects of decline in aquatic ecology (Karr, 1991; USEPA, 1996c; Yoder *et al*, 1996).

Cause and effect relationships are desired but may not be readily discerned or identified through our current methodology. This will be more true particularly for simplified monitoring programs that might rely on a single endpoint (Chapman, 1995). This example and other efforts do indicate however how GIS can be used to assist assessment of land use/cover and agricultural impacts to water quality and biodiversity (Osborne *et al*, 1988; Richards *et al*, 1994b).

The relative complexity of the hydrologic-ecological system will continue to impede our ability to link sources of “nearby” stress directly to degraded ecology instream. Studies that fail to account for hydrologic-hydraulic watershed functions at any given site may underestimate the far-field dominance of some stressors, such as AML-AMD. Ecological stress imparted by man is often controlled by activities upstream, and in most cases due to activity somewhere on the landscape, and not in the channel itself (Dunne *et al*, 1978; Leopold *et al*, 1964).

In addition to aquatic organisms, biota affected by AG or AML also may include wildlife populations and human populations at risk due to flooding or due to potential exposure to heavy metals that can occur at lower flows (Leopold *et al*, 1964; Nordstrom, 1982; USDA, 1985). Economic losses associated with impacted ecology include reduced recreational and economic potential incurred by the presence of large tracts of AML/AMD, and losses attributable to the associated risks of flooding (Shaffer, 1989; Smith *et al*, 1998; USDA, 1985).

These are all good reasons to better understand how AG and AML activities relate to watershed function and instream conditions. From the viewpoint of future watershed managers, functions or “ecological services” provided by the watershed may be viewed as points of final control (Cairns *et al*, 1991, 1996; NRC, 1992). Thus it is implied we will value what benefits us.

2.2.3 Connecting Hydraulics and Aquatic Habitat

The growing importance of connecting hydraulics to ecology is evident (Donigian *et al*, 1995). The following habitat model description illuminates how quantitative descriptions of physical habitat may define biotic production. Physical habitat considerations are key to success in understanding aquatic ecology and its decline under any circumstance. One example points out the rise in contemporary microhabitat measurement activity, where for example in-depth velocity profiles can be measured to assess fish habitat preferences (Beebe, 1996). Also developed to assess ecology was the fisheries based Physical Habitat Simulation Model, PHABSIM. The model directly relates hydraulic features of a reach to biotic production.

PHABSIM defines a method for evaluating the availability of physical microhabitat in various streams under different hydraulic scenarios (Wesche *et al*, 1980). This is a physical model, and does not directly evaluate chemical impacts. The computer model is based generally on the Instream Flow Incremental Method (IFIM), and can incorporate flow depths, velocity, substrate and cover preferences of fish. The IFIM two-prong approach includes hydraulic and habitat simulation at a cross-section point. The model can develop a stage-discharge relation based on any one of several hydraulic models such as power-function curve-fitting with stage-discharge data, using Manning's equation, or using a standard step backwater approach (Wesche *et al*, 1980).

In IFIM/PHABSIM use is made of "electivity" curves relating optimal habitat conditions for various combinations of velocity, depth, substrate, and cover. The later versions of the model can also evaluate other features of hydraulic geometry for instance usable-area, or wetted perimeter (Wesche *et al*, 1980). Weighted criteria may be further developed by subdividing a cross-section transect and applying the weighted criteria by segment, and later recombining on a per unit width basis (Bovee *et al*, 1977). Success of these species specific ecological assessment type models, which have typically been applied on a reach by reach basis only, are obviously dependent on the validity of the habitat-suitability curves employed.

The PHABSIM model may also be over-parameterized, leading to excessively large model uncertainty, particularly between users (Wesche *et al*, 1980). Other drawbacks in applying PHABSIM on watershed-scales are that habitat suitability curves are costly and may need to be developed site by site, species by species, involving a great deal of data collection (Wesche *et al*, 1980). Data and application experience may be limited for developing usability curves for non-Salmonid species and benthic invertebrate (Wesche *et al*, 1980). The model is still one of the few available that attempt to relate hydraulics directly to biodiversity in a simulation setting.

2.2.4 Pursuing Connectivity of Hydrology, Hydraulics, and Aquatic Habitat

Demand for integration of user-friendly basin models such as the Hydrologic Simulation Program – Fortran (HSPF) may eventually lean to ecology. This will occur in response to a need to facilitate more indepth simulation of ecological and ecotoxicological conditions (Cairns *et al*, 1991, 1996; Donigian *et al*, 1995). One of the more interesting generalized questions is how do we best bring together classical hydrologic simulation and classical ecological risk assessment?

Increases in demand for "integrative" type assessments will continue to follow the rising demand and value we place on clean water, controlled erosion, and flourishing biodiversity. Perhaps future investigations will arrive at some combination of applications like IFIM/PHABSIM and HSPF via a modeling platform like BASINS that can also integrate Geographical Information Systems (GIS) (Aronoff, 1993; Donigian *et al*, 1995; USEPA, 1996b; Wesche *et al*, 1980).

Watershed management is becoming more necessary for instance in limiting man's activity on the landscape where this activity might otherwise impact water intake plants, navigation, or recreation potential (Bonta *et al*, 1992; Broner *et al*, 1996; Curtis, 1979; Kershner, 1997; Lenat, 1984; Thomann *et al*, 1987; USDA, 1983, 1985). Increasing focus on water resources management will result in additional recognition given to aquatic ecology and the rising value society continues to assign to it (Cairns *et al*, 1996; Johnson, 1995; Minshall *et al*, 1985; NRC, 1992; Suter, 1998b; USDA, 1998; Vannote *et al*, 1980; Waters *et al*, 1995). A convergence towards integral study of hydrology and ecology, termed here as hydroecology, will then likely prevail in watershed management.

2.3 INTEGRATED ECOLOGICAL RISK ASSESSMENT

A logical breakdown of integrated landscape assessment problems may follow field, subwatershed and watershed-scale boundaries, upwardly progressing to basin-scale and regional-scale studies, depending on the concern (Aronoff, 1993; Bedient *et al*, 1992; Dunne *et al*, 1978; Linsley *et al*, 1975; Singh, 1995). We also know to varying degree that water quality, sediment quality, and biodiversity are intimately related (Chapman, 1990; Burton *et al*, 1992, Nelson *et al*, 1992; Rosenberg *et al*, 1986, 1993; Powers *et al*, 1992; USEPA, 1992). Most ecological studies conducted to this date though may be limited in some way in their ability to monitor or model the holistic watershed system (Burton *et al*, 1992; Cairns *et al*, 1996; Chapman, 1995; Singh, 1995).

Representative elements of ecology, an insect or clam per se, can now gain our focus as engineers (Cherry, 1996; OEPA, 1989). Biometrics can offer assimilative indication of water quality and can be used to measure overall system health (Burton *et al*, 1992; Cherry *et al*, 1979a, 1979b; Chapman *et al*, 1992; Chapman, 1995; Rosenberg *et al*, 1993; Shirazi *et al*, 1981). Understanding biotic condition in a water body provides in one sense a baseline measure of water quality that is sensitive and integrates anthropogenic influences (USEPA, 1996c). Biometrics may also be used to directly measure valued ecological components of a system placed under widespread management (Miller *et al*, 1988; Munkittrick *et al*, 1998; Richards *et al*, 1994b; Suter, 1998a; USEPA, 1992, 1998).

Of important discovery in the last decades is that aquatic environmental condition is largely determined by the current and past activities of the landscape its drains. (Fetter, 1993; Leopold *et al*, 1964). While this may now seem obvious, we are just beginning the needed work today to attain enhanced quantitative descriptions that eventually will relate landscape variables like %Strip Mines to variables describing water quality or quantity, and of more interest lately, biodiversity (Richards *et al*, 1994b, 1996; USDA, 1998; USEPA, 1980). It may be recorded eventually that this was the timeframe in which humankind learned how to reclaim nature from the systemic damage to landscape wrought after centuries of social progress. As such, we realize today that the existing risk assessment frameworks are in need of improvement. Associated field applications and analysis have been limited from the viewpoint of capturing all facets of watershed modeling, hydrology-hydraulics, or holistic system organization and analysis. This is truer for systems with multiple stressors (Cairns *et al*, 1991, 1996; Klump *et al*, 1998; Suter, 1998a; USEPA, 1998).

To explain biotic variation, we may find an ever increasing need to look at how biological parameters respond to the landscape through physically based techniques like simulation and land use/cover change analysis (Donigian *et al*, 1995; Higgins *et al*, 1987; Xinhao *et al*, 1997; Yoder *et al*, 1996). With our understanding of agricultural impacts limited, meaningful watershed assessment is further delayed by a lack of understanding and research dealing with the impacts of AML upon water quality and ecology (Babendreier *et al*, 1997; Cherry *et al*, 1995; Daniels *et al*, 1995; Nordstrom, 1982; OEPA 1991a, 1991b; USDA, 1985).

Conducting an integrated ecological analysis of a given watershed can provide an ability to evaluate all of the various elements, processes and functions within a watershed. One may conceptualize a watershed assessment using seven key steps (Cherry *et al*, 1999; Kershner, 1997; Suter, 1998a; USEPA, 1992, 1996c, 1998):

(1) Watershed characterization,
(2) Identification of issues and key questions,
(3) Documentation of current conditions,
(4) Description of reference conditions,
(5) Identification of objectives,
(6) Summary of conditions and determination of causes, and
(7) Recommendations.

2.3.1 IBI Scores in Watershed Assessments

In the future, an increased focus on water quality and quantity will align with increasing needs to quantitatively relate chemical and physical watershed variables (Bicknell *et al*, 1993, Donigian *et al*, 1995; USEPA, 1991, 1996; Xinhao *et al*, 1997). Added to this will be the existing desire to directly link hydrology and landscape to ecological biocriteria like IBI and ICI scores (Miller, 1988; Richards *et al*, 1992, 1994b). IBI scores, Index of Biotic Integrity value fish populations and represent a composite of 10 to 12 variables covering species richness and composition, trophic composition, and abundance and condition (Allan, 1995; Karr, 1981a; Karr *et al*, 1986; Miller *et al*, 1988). ICI refers to the benthic macroinvertebrate community index and is a construct of the State of Ohio (OEPA, 1989). IBI scores are defined in Table 2.1.

For the IBI scoring, each variable at each site is rated on a discrete scale of 1, 3, or 5. Variables 6, 7, 11, and 12 are negatively correlated in the rating where in general a large IBI score indicates excellent biotic condition. At each site scoring is supposed to be made relative to a consistent framework for drainage area and zoogeographic region, and may require a good deal of local calibration (Allan, 1995). Thus the IBI are drainage area dependent and, in similar manner, stream order dependent. Appreciation of scale effects is required in assigning a value in each case, but this may be a qualitative, subjective process.

Table 2.1 - Index of Biotic Integrity (IBI) Score
(Allan, 1995; Karr, 1981a; Karr *et al*, 1986; Miller *et al*, 1988)

Species Richness and Composition	
1.	Total # of Native Fish Species
2.	# and Identity of Darter Species (e.g. benthic species)
3.	# and Identity of Sunfish Species (e.g. water column species)
4.	# and Identity of Sucker Species (e.g. long-lived species)
5.	# and Identity of Intolerant Species
6.	% of Individuals as Tolerant Species
Trophic Composition	
7.	% of Individuals as Omnivores
8.	% of Individuals as Insectivorous Cyprinids
9.	% of Individuals as Piscivores (e.g. top carnivores)
Fish Abundance and Condition	
10.	# of Individuals in Sample
11.	% of Individuals as Hybrids or Exotics
12.	% of Individuals with Disease or Deformities

As Karr points out (1986), species richness and composition can vary substantially with stream size and region. Without quantitative approaches that address spatial dependence, the methods lack some objectivity. A concern may exist that spatial dependence effects can easily be lost in standardized reporting or ignored during assessment and generation of subscores or the final site IBI score. Hydrology, geology, fluvial geomorphology, and pollutant input can have significant effects on community assemblage both at regional-scale (Hughes *et al*, 1994; Richards *et al*, 1996; USEPA, 1980) and local-scale (Richards *et al*, 1994b).

2.3.2 ICI Scores in Watershed Assessments

Invertebrate Community Index (ICI) scores are created to value benthic macroinvertebrate populations in a somewhat similar manner as IBI scores (OEPA, 1989; Yoder *et al*, 1996). ICI scores were originally developed from in-house knowledge developed through years of work in Ohio. The index of macroinvertebrate assemblage was adopted from Karr's approach that was used to develop the IBI index for fish. The ICI and IBI have significant application potential in southeastern Ohio due to the availability of historical data and system wide as well ecoregion specific biological and chemical reference data (OEPA, 1989, 1991a, 199b; USEPA, 1988; Yoder *et al*, 1996). Quantitative physical data, both from a viewpoint of availability and efficacy, may be limited for most ecological risk assessments (Cherry *et al*, 1995; Richards *et al*, 1994b, 1996; Wesche *et al*, 1980; Yoder *et al*, 1996). Assessment of biotic integrity, through either fish (IBI) or insects (ICI), is a far more strongly embraced framework today, and imparts a holistic approach to measuring biological response from a wide array of chemical and physical stresses. A competing need will always remain though in equating these concepts to physical and chemical variables under the control of land use managers.

The ICI scores are usually comprised of 10 structural community variables where the first 9 are based on data collected via quantitative Hester Dendy sampling techniques that use an artificial substrate to sample *in situ*. The sampling devices are collected and a set of metrics may be developed per unit area of habitat. The 10th variable is based on qualitative dipnet sampling instream which in Ohio attempts sample collection at representative riffles, runs, and pools in a given reach (OEPA, 1989). A limitation in the ICI quantitative approach is that small watersheds (e.g. <10 km^2) and subsheds subject to massive sedimentation may not be amenable to measurement. In such cases, a qualitative approach would typically still be advantageous and advocated in assessing multiple subsheds. The ICI construction is defined I Table 2.2.

Table 2.2 – Invertebrate Community Index (ICI) Score
(OEPA, 1989)

1.	Total # of Taxa
2.	Total # of Mayfly Taxa
3.	Total # of Caddisfly Taxa
4.	Total # of Dipteran Taxa
5.	% Mayflys
6.	% Caddisflies
7.	% Tribe Tanytarsini Midges
8.	% Other Dipterans and Non-Insects
9.	% Tolerant Organisms
10.	Total # of Ephemroptera, Plecoptera, and Trichoptera (EPT) Taxa

The ICI variables are scored 0, 2, 4, or 6 points at each site relative to a consistent framework dependent on drainage area, ecoregion, and site condition. The point system evaluates a given sample against a sample of 247 streams in Ohio representing a wide range of potential and actual instream conditions. Ohio has four distinct ecoregions subdividing the state (Hughes *et al*, 1994; OEPA, 1991a; Omernik, 1987; USEPA, 1988). Thus, like the IBI, the ICI are also drainage area dependent, but this may again be a qualitative, subjective rendering. Similar concerns may exist that spatial dependence effects can be lost in reporting or ignored during generation of scores.

2.3.3 Uses of Biocriteria

Variables such as IBI and ICI scores representing fish and benthic macroinvertebrate, respectively, are favored by ecologists for their lumped, integral simplicity (USEPA, 1996c). Based on the watershed conditions incorporated by these scoring approaches, IBI and ICI scores are presumably under strong influence by both AG and AML activities. Of interest in describing and solving the problem of how to assess AG and AML impacts is learning how land use/cover patterns can be better related to engineering variables (e.g. %aereal disturbance mean particle size, washload rates, etc.) and ecological variables (e.g. macroinvertebrate total taxon richness, EPT abundance etc.).

The index rating systems described for IBI and ICI, carried out by practiced examiners can be potentially successful in relating both AG and AML land use measures to IBI scores (Allan, 1995; Burton *et al*, 1992; Cherry *et al*, 1995; Richards *et al*, 1994b; Roth, 1994;

Steedman, 1988). Several discussions are available relating various constructions and applications of biotic indexes (Lenat, 1993; Miller *et al*, 1988; Steedman, 1988). OEPA in fact uses both IBI and ICI scores together in assessing overall biological use-attainment standards for streams throughout Ohio (Cherry *et al*, 1999; OEPA, 1989, 1994; Yoder *et al*, 1996).

Clean water and diverse ecology are now viewed as scarce resources, each highly valued. Measures of water quality and diverse ecology are likely to be positively correlated. Improvement objectives and related programs for assessing water, sediment, and ecology will therefore likely share common themes in watershed management (Klump *et al*, 1998). To describe the ecosystem and for example predict IBI and ICI scores throughout a watershed, many variables may need to be evaluated collectively, and in some manner integrated into a consistent and coherent modeling framework (Donigian *et al*, 1995; Singh, 1995).

2.3.4 Bioassessment – Components of Biocriteria

A major distinction has been formally set apart in ecological risk assessment in streams and small rivers. It has been recognized that there is a need to better distinguish between the attributes of the system we intend to protect (e.g. final assessment endpoints) and the attributes that we can actually measure (USEPA, 1996c). As an alternative construction, Karr's evolved integrity assessment approach has yet another more broadly defined framework offered more recently and it continues to be refined (Karr, 1991; USEPA, 1996c). The description is referred to as "components of biological integrity" and contrasts the elements of the biosphere with its processes. It is described by elements in Table 2.3.

Table 2.3 – Evolving Components of Biological Integrity (USEPA, 1996c)

Elements	**Processes**
Genetics	Mutation, recombination
Individual	Metabolism, growth, reproduction
Population/Species	Age specific birth and death rates; Evolution/speciation
Assemblage (community & ecosystem)	Interspecific interactions; Energy flow
Landscape	Water cycle; Nutrient cycles; Population sources and sinks; Migration and dispersal

Three major points were made recently by USEPA concerning overall components of biological criteria. First, success in protecting "biological integrity" depends on the development of measurement endpoints that are highly correlated with assessment endpoints. Secondly, efforts to assess these endpoints must be broadly based where this facet ranks higher than the actual choice of specific biocriteria used (assuming they are holistic and well defined). Finally, the best approach to assessing biological integrity is an integrative one that combines assessment of the extent to which either elements or processes of integrity are altered. (USEPA, 1996c).

There is likely to be continued expansion and generalization of the critically important underpinning concept of biodiversity. A general push exists which is expanding the use of these systems measuring biotic integrity to cover wider geographic areas (Oberdorff *et al*, 1992; OEPA, 1989, 1991a, 1991b). This is being expanded by efforts like the Ohio EPA's continued move to examine benthic macroinvertebrate assemblage within the context of fish and insect integrity along with efforts by others states like Nebraska (Bazata, 1991; OEPA, 1989; USEPA, 1989, 1996c). Ohio is one of the states farthest along in these types of efforts. This also coincides nicely with Ohio's advanced and active GIS community. Noted briefly above and to be discussed further below, Ohio makes novel use of both IBI and ICI scores in combining them into "use-attainment" standards for warmwater habitat and exceptionally warmwater habitat (OEPA, 1989, 1991a, 1991b; Yoder *et al*, 1996). For these reasons, the selected study area, Leading Creek Watershed located in southeastern Ohio, is ideal for further examining engineering relationships between biology and the landscape.

2.3.4.1 Bioassessment – Monitoring Aspects of Human Impact

With respect to biological criteria, a breakdown of the natural system can be first distinguished along lines of external and internal variables (Karr, 1991; USEPA, 1996c), where separation between internal and external variables is drawn along the riparian corridor. One aspect of this may include seasonal sampling that accounts for temporal variation in biotic response (OEPA, 1989; USEPA, 1996c). Weather/climate, terrestrial-environment, and land-use define the external aquatic system. Terrestrial-environment would include geology, topography, soil, and vegetation. This conceptual framework can be used to determine which biota should be evaluated.

2.3.4.1.1 Key Internal System Variables

Karr identifies five key internal variables that are subject to anthropogenic control and influence (Karr, 1991; USEPA, 1996c).

- ***Water Quality –*** Temperature, turbidity, dissolved oxygen, acidity, alkalinity, organic and inorganic chemicals, heavy metals, and toxic substances.
- ***Habitat Structure –*** Substrate type, water depth, current velocity, spatial and temporal complexity of physical habitat.
- ***Flow Regime –*** Water volume, temporal distribution of flows.
- ***Energy Source –*** Type, amount, and particle size of organic material entering stream, seasonal pattern of energy availability.
- ***Biotic Interactions –*** Competition, predation, disease, parasitism, and mutualism.

Yoder and Rankin (1996) have also more recently expanded upon elements comprising each of the five key internal variables.

2.3.4.1.2 Key Biological Components and Sample Attributes

At least four components of biota that may need to be assessed and further described along with the following sample attributes (Karr, 1991; USEPA, 1996c):

- ***Community Structure –*** Species richness, relative abundance, including the extent to which one or a few species dominates.

- ***Taxonomic Composition –*** Identity of the species that make up the biota; sensitivity or intolerance, rare/endangered taxa, other key taxa.

- ***Individual Condition –*** Health status of individuals in selected species; disease, anomalies, contaminant levels, and metabolic rates.

- ***Biological Processes –*** Rates of biological activities across the biological hierarchy (genes to landscape); trophic dynamics, productivity, predation and recruitment rates.

2.4 WATERSHED STRUCTURE AND SCALE

A direct consequence of integrated watershed assessment is that it involves collection of a significant amount of “basic” data and “specialized” data, spanning multiple disciplines and interests (Bedient *et al*, 1992; Linsley *et al*, 1975). Large studies as such need to be placed within a hydrologic-hierarchy for information management consistent with contemporary watershed investigation concepts (Aronoff, 1993; McCammon, 1998; Singh, 1995).

Data evaluation can occur on several structural levels where a base GIS can today typically be developed at 1:24000 scale using either simple or sophisticated digital data generation techniques (Arnold *et al*, 1995; Aronoff, 1993, Donigian *et al*, 1995). Some of the more sophisticated approaches can fully exploit aerial photography, remotely sensed images such as those provided by Landsat5 (30m resolution), and more currently by Landsat7 satellite and SPOT satellite data (10m resolution) (Aronoff, 1993). Resolution indicates the smallest recognizable (square) unit of area that can be assigned as an element of some feature class, such as classifying a square polygon as being urban or agriculture (Aronoff, 1993).

For hydrologic simulation models, the effectiveness of remotely sensed data has been shown (Bicknell *et al*, 1993; Donigian *et al*, 1995; USEPA, 1981). Use of equivalent or finer scale data from aerial photos and soil surveys may also be available to relate changes in land use/cover to changes in water quality (Fauss, 1992). Coarser-scale information can be gleaned from widely available (and free) government map series ranging in scale from 1:100,000 to 1: 2,500,000. These maps can have several uses, particularly at subbasin to regional scales, described further below (Richards *et al*, 1994b, 1996). The coarser frameworks such as scales of 1:100,000, however, may be unable to adequately identify or compare biological data collected at watershed and subwatershed scales (Aronoff, 1993; Richards *et al*, 1994b). Today, for example in Ohio, 1:24000 mapping is becoming increasingly more accessible at low or no cost to the user, though considerable expense may still be entailed in analysis of these geodata sets.

2.4.1 Hydrologic Units

The USGS's and the USDA's basic hydrologic unit organization and terminology include:

- Region,
- Subregion,
- Accounting unit (or basin),
- Cataloging unit (or subbasin),
- Watershed,
- Subwatershed,
- Subshed (or drainage), and
- Site.

The terminology used describes hydrologic unit naming conventions consistent with a nationally recognized identification system (McCammon, 1998; Seaber *et al*, 1987). The Leading Creek Watershed study site in southeastern Ohio falls within the [Ohio River]-[Upper-Ohio]-[Little-Kanawha]-[Shade-River] Cataloging Unit distinguished by the USGS (8-digit) Watershed ID = # 05030202. Ohio has four distinct ecoregions subdividing the state (Omernik, 1987; OEPA, 1991a, 1991b; USEPA, 1988), as well as its own watershed identification system. Ecoregional distinctions may not always necessarily coincide directly with hydrologic unit descriptions (Omernik *et al*, 1991). Resources may need to be viewed differently and water quality may need to be managed across regional lines.

2.4.2 Scale Effects

In the above framework, typical aspects of field-scale, landscape-scale, and watershed-scale approaches can be incorporated (Aronoff, 1993, Richards *et al*, 1994b, 1996; Singh, 1995). These may also be discussed in the framework of hydrologic simulation scales (Singh, 1995).

(1) Lab-scale,
(2) Hillslope-scale,
(3) Catchment-scale,
(4) Basin-scale,
(5) Continental-scale, and
(6) Global-scale.

On its base level, a study and underlying scale concepts will be defined by selecting a certain set of sites for data collection. Site selection in an integrated assessment environment is of obvious critical importance (USEPA, 1996c). Scale may be defined as the size of a hydrologic unit within which (we would normally assume) that the hydrologic response is homogeneous (Singh, 1995). Scale is a question of degree of homogeneity or, if you wish, degree of heterogeneity (Singh, 1995). With an established order and recognition of spatial effects, one may begin to infer conclusions about the character of a hydrologic unit, both inside and outside its effective domain. Subwatersheds in this way can be investigated in context of the larger watershed, assessing the aggregate subwatershed impact upon the mainstem for example. A study can conceivably be fully integrated with knowledge gathered throughout the watershed (Bedient *et al*, 1992; Linsley *et al*, 1975).

Of important notice is that many biological systems are scale dependent, where drainage area and to a lessor degree stream order may be a useful surrogate (Borcard *et al*, 1992; Cummins, 1983; Karr *et al*, 1986, 1991; Minshall *et al*, 1985; OEPA, 1989; Strahler, 1957; USEPA, 1996c; Vannote *et al*, 1980). Spatial dependence was found to be important in the discussion of *Corbicula fluminea* used in ecotoxicological *in situ* testing *(*Cherry, 1996; Babendreier *et al*, 1998a). A community desires a study that has digested this information and has arrived at a final set of easily understood conclusions and recommendations (USEPA, 1996c). The products of a work defining both a diagnosis and prognosis for an ecosystem should employ an objective, defendable, system-wide evaluation (Cairns *et al*, 1977; USEPA, 1991, 1996c). In completing any initial and then final analysis, a typical risk assessment may also conceivably need to reflect on the hydrology and ecology of larger units and ecoregions that might encompass the watershed (Cairns *et al*, 1996; USEPA, 1988, 1992, 1998).

A single study must gather data on various structural and functional levels of the ecosystem, both parental and internal (Vannote *et al*, 1980). Data must be appreciably organized, queried and then analyzed within an ecological risk assessment framework (Babendreier *et al*, 1998b; Suter, 1998b; USEPA, 1992, 1998). Any methodology chosen must afford a sound, technical and thorough inspection of the watershed's ecological and hydrological interaction, utilizing both quantitative and qualitative measures (Babendreier *et al*, 1998c; McKeon *et al*, 1988; NRC, 1992; USEPA, 1991). A sufficient level of knowledge about cause and effects related between sources of stress and variation in aquatic biodiversity is equally desired. This is especially true if we apply numerical standards that may be used to gage need or success of landscape or stream restoration projects (USDA, 1998; USEPA, 1996a, 1996c, 1999).

2.4.3 Defining A Watershed

A key term related to watershed assessment is geomorphology, defined as the study of the form and structure of the earth's surface, both past and present. A geomorphic cycle is synonymous with erosion (Leopold *et al*, 1964). Fluvial geomorphology is the study of rivers and streams dissecting this "living" surface. To provide a consistent framework for discussion, the following terminology describing hydrologic units is also introduced to better define what the word "watershed" means.

The word "watershed" can mean many things to many people. For example, the term "watershed" may refer to a community-based organizing committee associated with flood control management in the State of Ohio. With more loss of generality, watersheds have also been popularly defined lately by catch-phases such as *"if you live anywhere, you live in a watershed"* (USEPA's Surf Your Watershed website, 1999). These phrases have begun to popularize our understanding that hydrology and hydraulics do largely determine environmental condition instream. This can also be summarized by the sticker phrase "we all live downstream" which graces many automobile bumpers today.

The term watershed previously had been more succinctly defined as a divide separating one drainage area from another (Black, 1997; Chow, 1964; Leopold *et al*, 1964; Linsley *et al*, 1975). For purposes of this study, the word "watershed" and terms like it are delineated further to impart a needed sense of spatial scale between various hydrologic units (McCammon, 1998; Seaber *et al*, 1987). The latter contemporary term used, hydrologic unit, may be a more appropriate object of Chow's original definition of the term watershed. For purposes of hydrological and ecological risk assessment, a proper definition demands multiple levels to address critical issues of scale. A consistent framework is needed to evaluate watershed facts and to identify stressor impacts upon biotic and abiotic conditions (Aronoff, 1993; Bedient *et al*, 1992; Cairns *et al*, 1991; Linsley *et al*, 1975; Minshall *et al*, 1985; Vannote *et al*, 1980).

2.4.3.1 Contemporary Watershed Identification System

The Leading Creek Watershed study site is identified by the following conceptual and standardized drainage basin classification system. This system, combined from USGS and others, identifies several layers of "hydrologic units". By design, this terminology can be used to cross several spatial scales encountered in any study of one contiguous area, no matter how large or small that area may be.

- USGS Hydrologic Region: *Ohio River* (05)
- USGS Hydrologic Subregion: *Upper-Ohio River* (03)
- USGS Hydrologic Accounting unit (basin*): Little Kanawha River* (02)
- USGS Hydrologic Cataloging unit (subbasin) *: Shade River* (02)

For purposes of further defining units within the Shade River Subbasin and Leading Creek Watershed, descriptions are also given for the following conventions adopted (Seaber *et al*, 1987; McCammon, 1998):

- Hydrologic Watershed: *Leading Creek*
- Hydrologic Subwatershed: *Primary Tributaries of the Mainstem*
- Hydrologic Subshed (or drainage) *: A Portion of Some Subwatershed*
- Hydrologic Site or station: *A Hydrologic Unit's Outlet Point (referring to cross-section attributes of the uniquely defined hydrologic unit upstream)*.

The US Geological Survey (USGS) has compartmentalized the basic hydrologic units of the United States (Seaber *et al*, 1987). This arrangement allows for definition of smaller units within larger units, much like a set of mixing bowls in your kitchen (McCammon, 1998). To improve communication in watershed management concepts, the USGS developed this standardized definition set, used here where feasible, in referring to and establishing the interconnection and relative sizes of various catchments. The specific portions of Leading Creek Watershed found within larger categorized hydrologic units can be seen in Figure 2.1 (USGS Classification) and Figure 2.2 (Ohio EPA Classification). Figure 2.2 also classifies OEPA's approach in monitoring network design. A site is given as a hydrologic unit's outlet point (or catchment outlet) and refers to attributes of the uniquely defined upstream hydrologic unit. Hydrologic units would be described by polygons, and the streams and rivers would be described by lineal elements.

USGS Hydrologic Region: **Ohio River** (05)
USGS Hydrologic Subregion: **Upper-Ohio River** (03)
USGS Hydrologic Accounting unit (basin): **Little Kanawha River** (02)
USGS Hydrologic Cataloging unit (subbasin) : **Shade River** (02)

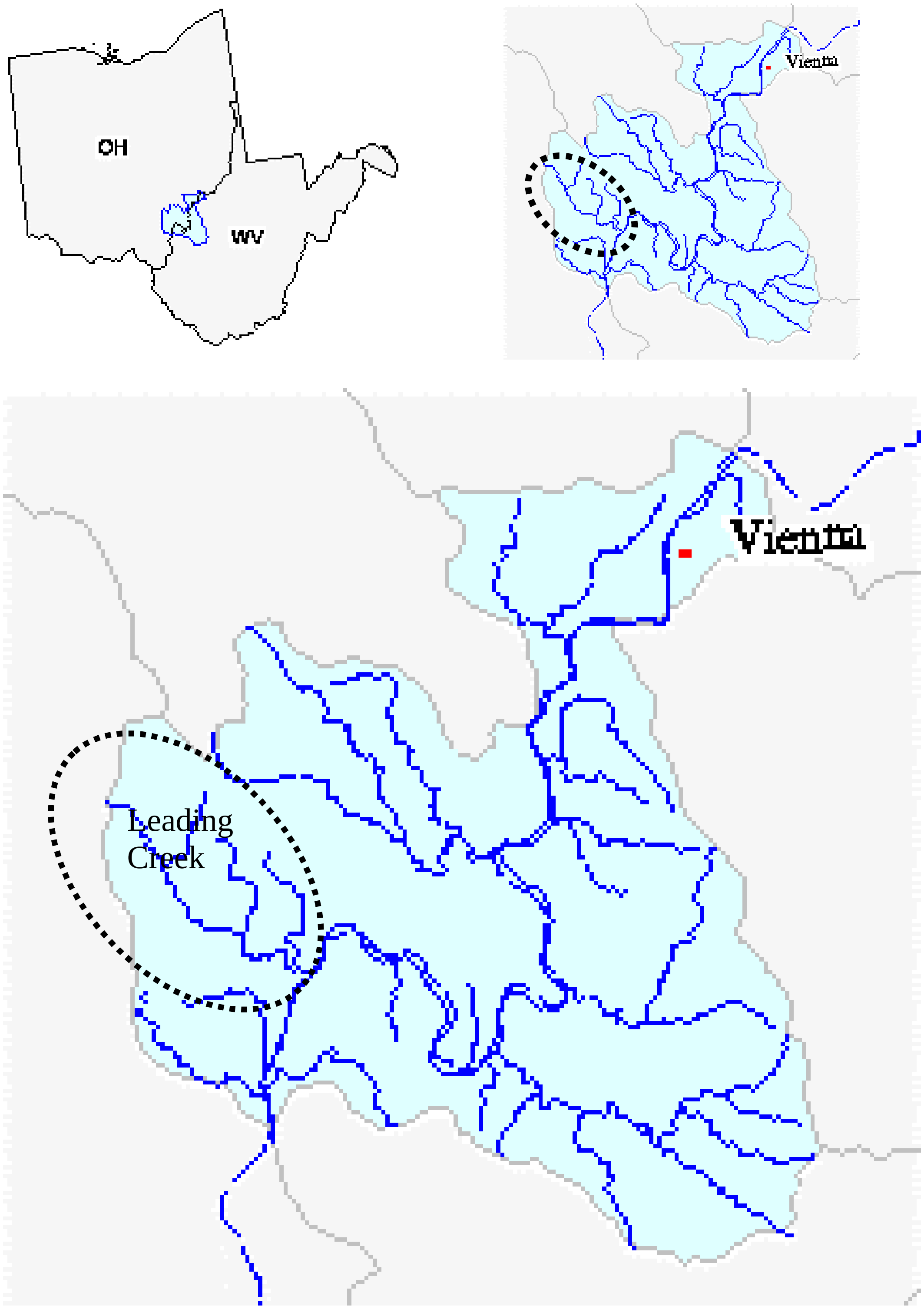


Figure 2.1 - USGS Contemporary Watershed Identification System

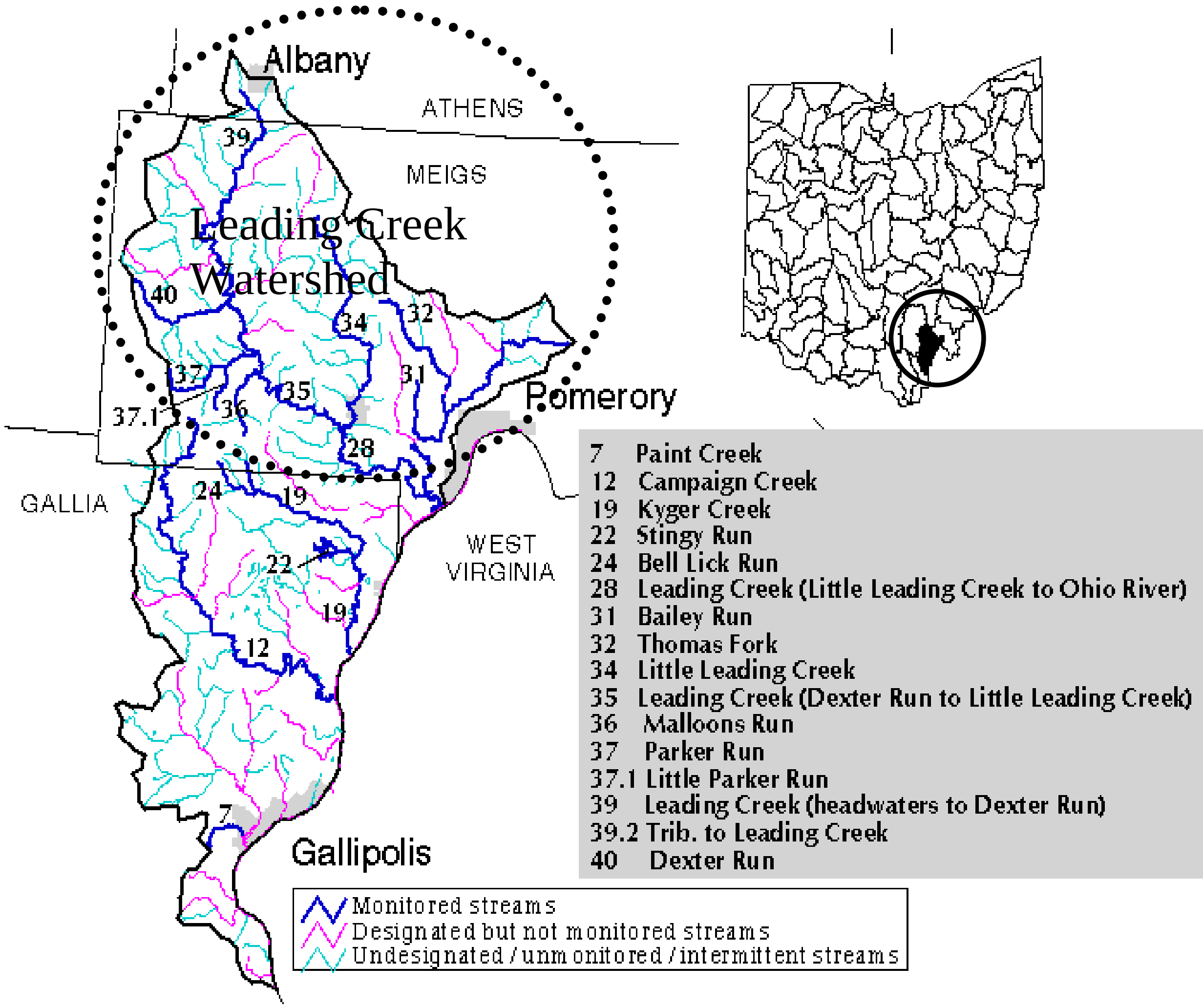


Figure 2.2 - Ohio EPA (OEPA) Contemporary Watershed Identification System

United States was first divided by the USGS into the top four levels noted: regions, sub-regions, accounting units (or basins), and cataloging units (or subbasins). The smallest unit recognized by USGS to date is the cataloging unit. With 2150 cataloging units in the U.S., most drain greater than 700 square miles (McCammon, 1998). The largest hydrologic units in the U.S. are "regions" of which there are 21 total, 18 in the conterminous U.S. (e.g. the Ohio River Basin, noting the confusing use of the word basin ahead of time). There are 222 subregions and 352 accounting units (McCammon, 1998). Regions typically represent major rivers of the U.S. or a series of rivers draining a similar region (e.g. Missouri and Texas-Gulf regions).

2.4.3.2 Other Hydrologic Unit Coding Systems

Each hydrologic unit is identified by its unique hydrologic unit code (HUC) consisting of two to eight digits based on the four levels of classification in the hydrologic unit system. Some refer to the base USGS unit, the cataloging unit, as a "watershed", for example in USEPA's Surf Your Watershed website. Some classification systems have also extended the 8 digit code to 16 digits in order to address identification of more refined concepts described above like "watershed", "subwatershed", and "subshed" (McCammon, 1998).

Classification of land drainage is not always consistent with the established USGS convention. An example of such classification is provided by Ohio EPA's *"Find Your Watershed – website"*. In its organization of state streams, Ohio developed a *Principal River or Stream Group Numbers* naming system. In this system, Leading Creek falls within Group 29, Southeast Tributaries of the Ohio River – Lower Raccoon Creek. Group 29 identifies drainages from the Northern/Western shore of the Ohio River, starting at Leading Creek in Middleport, Ohio, through Kyger and Campaign Creeks to just below Paint Creek near Gallipolis, Ohio.

To avoid confusion in comparing various watershed identification systems for Leading Creek, note that:

- ♦ Raccoon Creek's drainages are not found in Group 29 of the Ohio EPA's system,
- ♦ The Ohio EPA's website currently shows Group 28 to be named after the Shade River and Leading Creek, where Leading Creek is part of Group 29, and
- ♦ OEPA's Group 29 dissects the encompassing USGS cataloging unit for the Leading Creek area, which is the Upper-Ohio Shade (USGS Watershed ID = 05030202).

Some confusion may be potentially associated with typographical errors in the source information, or may otherwise need to be corrected, or the identification may be correct. Despite their differences, both identification systems, by USGS and Ohio EPA, are useful to their respective programs. For future work, the USGS convention is always recommended at a minimum. It does not appear that a consistent 16-digit identification of the Leading Creek Watershed and its tributaries has been identified to date. The discussion confirms the competing needs of management that may be politically determined for instance. Reconciling various approaches also points out the underlying structural reality of the ecological and hydrological systems being managed (Hughes *et al*, 1994; May *et al*, 1998; Omernik *et al*, 1987, 1991).

2.5 FLUVIAL GEOMORPHOLOGY

2.5.1 Idealized Fluvial System

A background discussion on geomorphology and an idealized fluvial system is provided next. The idealized system describing the interaction of morphologic units of hillside, channel, and mouth/delta are taken from Schumm (Diplas, 1997; Schumm, 1977). Also provided are the corresponding fluvial system units of production, transfer, and deposition, respectively. The three equivalent zones defined by the fluvial units are intuitively related. The first two cover in some manner hydrology and hydraulics, respectively, and a third deposition zone introduced is the ultimate storage unit holding the final fate of sediment run-off from the land surface. For Leading Creek Watershed, the fate of (most) water and sediments is the Ohio River, Mississippi River, and the Gulf of Mexico.

Under normal watershed circumstances, Zone 1 in Tables 2.4 and 2.5 would be classified as an area producing sediment and water, but having insignificant (readily available) storage. Naturally occurring stability generally increases, moving from Zone 1 towards Zone 3. This is a scale-dependent issue; a production element in one rendition or model of a watershed may be a transfer element in another model.

Table 2.4 - Morphology and Idealized Fluvial System (After Schumm, 1977)

Morphology Component	**Fluvial System Component**	**Fluvial Zone #**	**Area of Study**
Hillside	Production	Zone 1	Hydrology/Geomorphology
Channel-valley	Transfer	Zone 2	Hydraulics
Mouth/Delta	Deposition	Zone 3	Coastal Geology/Engineering

Table 2.5 - Conceptual Morphology and Idealized Fluvial System Variables
(After Diplas, 1997; Leopold *et al*, 1964; Schumm 1977)

Fluvial Zone #	**Fluvial Var. #**	**Fluvial Zone Variables**	**Variable Type**	**Comments**
Zone 1	**1**	Time	Independent	
	2	Initial Relief	Independent	
	3	Geology	Independent	Lithology & Structure
	4	Climate	Independent	
	5	Vegetation	Dependent	Determined by variables 3-4; type - density.
	6	Relief	Dependent	Determined by variables 1-5. Conceptually the volume of system above the base level.
	7	Initial Hydrology	Dependent	Determined by variables 1-6; Run-off and sediment yield per unit area.
	8	Drainage Morphology	Dependent	Variable 7 determines drainage network, constrained by the variables 1-4, 6.
	9	Hillslope Morphology	Dependent	Variable 7 determines hillslope, constrained by variables 1-4, 6.
	10	Final Hydrology	Dependent	Determined together by variables 7, 8, & 9. Discharge of sediment & water to Zone 2.
Zone 2	**11**	Channel-Valley Morphology	Dependent	Determined largely by volume of water and sediment. Channel morphology & sediment characteristics.
Zone 3	**12**	Deposition-Mouth Morphology	Dependent	Determined largely by volume of water and sediment. Depositional system morphology and sediment characteristics. (a.k.a. mouth/delta)

2.5.2 Conceptual Fluvial Model Elements

An important concept in landscape hydrology is feedback. Independent variables 1-4 in Table 2.5 determine, in sequence, vegetation and initial relief, which in turn sets up an initial hydrology. The key feedback step is hillslope and drainage network development that continually redefines itself by redefining “initial” and “final” hydrology (Diplas, 1997; Leopold *et al*, 1964; Linsley *et al*, 1975). Hillslope and drainage network morphologies themselves strongly influence production of water and sediment, which in turn redefines variables controlling hillslope and drainage patterns. Hillslopes represent that part of the landscape included between the crest of hills and their drainage lines (Leopold *et al*, 1964).

In the idealized system, the hillslope/drainage network transitions to a channel-valley zone comprised of the main transport channel(s) and its lower terrace floodplains. On shorter time scales, certain parts of the system may reach effective equilibrium (Diplas, 1997). Of the two elements or zones important to Leading Creek, hydrological drainage networks (Zone 1) are often characterized by less storage and less stability. Hydraulic network behavior or Zone 2 is characterized by increased storage and increased stability (Diplas, 1997; Schumm, 1977). For Zone 2, "mainstem" can be a useful description for watershed-scale studies and imparts the primary function of transfer (Black, 1997; Bedient *et al*, 1992; Linsley *et al*, 1975).

Imparting a one-directional (gravity-induced) flow of water and sediment in the idealized model, the three morphological units are reasonably distinguishable from one another as exchange points of sediment and water. The hillside (hillslope and drainage network) flows to the channel-valley which flows out to the mouth-delta. This model is consistent in general with describing the overall processes relating to erosion of the landscape that are intimately driven by the hydrologic cycle (Bicknell *et al*, 1993; Fetter, 1994; Leopold *et al*, 1964)

Hydrology may be defined in the conceptual model used here as the sediment and water production and discharge of a fluvial system (Bedient *et al*, 1992; Linsley *et al*, 1975; Schumm, 1977). The land produces the sediment and water. Run-off acting upon the rock and soil regimes characterize hillslope and drainage networks (Diplas, 1997). The networks in turn equilibrate the sediment and water, yielding a final hydrology (Diplas, 1997). Physically, the discriminating factor defining where hillside morphology ends and channel-valley morphology begins is not always clear. It is scale dependent and depends on how the underlying information is to be used or modeled (Singh, 1995; USEPA, 1991, 1998).

2.5.3 Conceptual Fluvial Model of Leading Creek

A specifically highlighted conceptual model for Leading Creek and its vantagepoints are offered below. The discussion is presented to discuss how one can classify significant features of a watershed according to Schumm's idealized conceptual framework. The study features will typically have to include, at the most basic level, definition of the dominant hillside hydrology, floodplain activities, and their relation to "main" channels. This discussion is based on assumptions of watershed-scale and subwatershed-scale monitoring that for example might be used in an assessment and improvement scheme for a 388km^2 watershed like Leading Creek.

For purposes of this study, the lower portions of the mainstem of Leading Creek (LC), below Mud Fork (above Dexter, Ohio), and lower portions of Little Leading Creek (LLC) and Thomas Fork (TF) tributary-stems were categorized as channel-valley units based on initial field data. The term "tributary-stem" will be used to refer to a mainstem trunk line on a designated tributary of the model mainstem, in this case Leading Creek. These transport channels move significant amounts of sediment and water with respect to further increases in local production downstream. Not coincidental, the stream network defined in Figure 2.1 accurately depicts the pathways of greatest transfer and indeed represents by and large the Leading Creek stream segments that were identified in preliminary field surveys.

Upgradient of the mainstem and 2 tributary trunk lines (LC, LLC, and TF), the definition of final hydrology is more interactive within drainages. Hillside morphological functions prevail with overland flow and infiltration discharging to watercourses (Linsley *et al*, 1975). In the fluvial model, hillside morphology plays its familiar role as a sediment production area. At a study scale of 1:24,000, upgradient features of the Leading Creek stream system could be generalized as part of the Hillside Zone #1, in Table 2.4 and Table 2.5. The Hillside Zone is conceptually comprised of normally dry land surface elements (or lowest order subsheds; including gullies, rills, and ditches), and a servant drainage network (or lower to higher order subsheds and subwatersheds of the mainstem).

2.5.3.1 Aspects of Subwatershed and Finer Scale Approaches

The analysis design captured by the Leading Creek study and its associated work plans was defined by its charge of evaluating 388km^2 of land within a short time frame. With a practical limit of monitoring only about 25-50 sites for various parameters, this necessarily defined a subwatershed-scale modeling approach. As described above, hillside morphology is typically characterized by minimal storage functions of readily available sediments, where channel-valley units are characterized by significant storage functions. To optimally model heterogeneous AML directly for drainage or field-scale remedies (e.g. an individual hillside), drainage-scale fluvial model applications could likely be more useful than subwatershed-scale analysis attempted here. This would appear to be prudent in accounting for relatively small and large individual contributions from each hillside area (Donigian *et al*, 1995; Gillmore *et al*, 1991; USDA, 1985). Such a model would be prohibitively expensive to calibrate at this scale

Not directly addressed by Schumm's idealized fluvial model, radically disturbed hillslope areas of surface mines may need to recognize an element of more readily available storage of sediment. This may even entail mass movement (Leopold *et al*, 1964). Normal land scour routines that evaluate rain-drop and sheet flow scour would need to directly address localized heterogeneity of these disturbed hillsides within Zone 1 (e.g. unreclaimed strip mines) and storage in Zone 2 (Bedient *et al*, 1992; Linsley *et al*, 1975). For example, using the Hydrologic Simulation Program Fortran (HSPF) model (Bicknell *et al*, 1993), this would require a fine-resolution approach which encompasses the variation in disturbance (e.g. mining practice) and reclamation status within Zone 1 (e.g. soil properties, cover properties, aereal extent, and landform slopes; scales of 1:24,000 or 1:15,000). A better understanding of longitudinal heterogeneity of sand bar storage along key subwatershed "channel-valley zones would also appear to have a significant potential to advance optimization of the field-scale problem at hand.

The increased need for spatial scale refinement in the case of AML impacts appears to be due to sand sedimentation, transported as suspended load from the hillside to the channel areas, and then down the stream system. Correction of acid mine drainage (AMD), to the degree it can be addressed, would also be given a better advantage in understanding the geological and hydrogeological sensitivity of each hillside, subshed, subwatershed, and watershed within the domain of Leading Creek (USEPA, 1980).

On all scales, the quality of modeling results depends directly on the quality of model input data (Singh, 1995; USEPA, 1991).

2.5.4 Mainstem and Primary Tributary Hydraulic-Transfer Reaches

The conceptual fluvial model of Leading Creek can be drawn along three lines of major changes in energy slope profile and major soil association distinctions in the system. The latter effect may be due to the relatively flat profile of bedrock dip in the region (Lucht *et al*, 1985). Keeping in mind a basic assumption that monitoring in this study was based on watershed-scale (i.e. reference comparisons) and subwatershed-scales (i.e. inter-watershed comparisons), major areas exhibiting dominant transfer functions may be practically viewed to occur in:

1) The Leading Creek mainstem, below Mud Fork,
2) Lower portions of the mainstem of Little Leading Creek, and
3) Lower portions of East Thomas Fork mainstem to its confluence with Leading Creek.

These segments, identifiable in Figure 2.1 and Figure 2.2, represent zones along which hillside frontslope morphology joins the more gently sloping channel-valley zone, itself draining to lower relief of the Ohio River (Leopold *et al*, 1964). Associated uplands of Leading Creek transition eventually to relatively homogeneous channel-valley morphologies. In this generalized classification, significant spatial scale effects may be evident between the three subwatersheds and their associated trunks. Leading Creek drains 388km^2, Little Leading Creek 66.3km^2, and Thomas Fork 80.4km^2. Drainage areas of subwatersheds studied in Leading Creek and reference watersheds are presented in Table A1 in Appendix A.

Without significant geomorphologic changes induced in part by mining, a revised designation of primary trunk-lines might otherwise have followed a simpler transition into the Chagrin (Cg) Soil Series, a silt loam dominating most channel-valley soil units in the lower watershed areas. Without the influence of AML-sand transport from Mud Fork, the Leading Creek mainstem area associated with dominant transport functions would begin farther south. The otherwise pre-disturbed trunk designation guided by soil association transition would start nearer to the Town of Dexter, Ohio, below Dexter Run, and would possibly exclude the Leading Creek drainage areas of Grass Run and Parker Run, down to just above Malloons Run above Langsville, Ohio. These effects indicate the influence that man can have on fluvial geomorphology and critical parameters that may determine biotic integrity (Leopold *et al*, 1964, USEPA, 1996c; Yoder *et al*, 1996).

2.5.4.1 Hillside Functions - Feeding the Mainstem

Among the three primary trunk lines identified, some differences in upland geology were also evident, from subwatershed to subwatershed area (Gillmore *et al*, 1991). “Upland” is incorporated here by definition as all of the crest-slope, mid-slope, and front-slope areas of a given hillslope morphological unit (Leopold *et al*, 1964). Upland would include ridgetops, very steep to steep to moderately steep hillside areas, and those areas adjoining channel-valley elements lying on moderately steep to gently sloping lands, including higher terraces adjoining the dominant valley floor (floodplain) profile (Gillmore *et al*, 1991, Lucht *et al*, 1985).

2.5.4.2 Mainstem Channel-Valley Function

Soils in the two primary tributary-stem zones of Leading Creek tracked the Chagrin-Nolan-Licking Association, where the former two soil-series in the association dominates most of the channel-valley unit (i.e. the exposed lower watershed valley floor). The major soil association encountered is relatively consistent across the channel-valley units of the three subwatersheds noted (LC, LLC, and TF). In general, the Nolan(No) soil series dominates higher elevations on the mainstem, and primarily the Chagrin(Cg) series dominates at lower elevations on the mainstem and primary tributary-stems of Little Leading Creek and Thomas Fork (Gillmore *et al*, 1991). These are all silt loam texture soil series (Lucht *et al*, 1985).

2.5.4.3 Mouth/Delta Functions - Receiving the Mainstem

The basis for assuming this fate for water, washload, and soluble load is based upon dilution, where Leading Creek makes up approximately 0.54% of the flow of Ohio River at Middleport, based on drainage area. The Ohio River channel itself functionally serves as a channel-valley unit for the larger Ohio River Basin (region). The regional hydrologic unit (203,940 sq. miles) is no longer a free flowing river. At Middleport, Ohio, the Ohio River bed sediments are expected to be in the fine sand to clay range. Based on induced deposition of sediments due to control of the Ohio River through locks and hydroelectric dams, the Ohio River must be periodically dredged at selected locations upstream of control points.

Larger sediments from Leading Creek are trapped somewhere above Gallipolis, Ohio, the nearest downstream lock facility (e.g. a water level control point affecting the mouth-waters of Leading Creek – e.g. backwater) (Bedient *et al*, 1992; Linsley *et al*, 1975). Sediments need periodic dredging over time to maintain minimum channel navigational requirements on the Ohio River. With some attenuation possible due to settling, significant portions of washload (fine materials <0.125mm) that discharge from Leading Creek continue to move downstream, along with soluble materials not precipitated (by definition materials <0.45 microns).

2.5.5 Summary of Conceptual Fluvial Model of Leading Creek

The model is conceptual only. The conceptual model is a tool that can be used to begin to describe Leading Creek, its structural elements and its attributes. It will be expanded throughout the remainder of the study to convey a consistent, familiar set of definitions and better understanding of system processes. At any given point in space, the model described here is rooted in the relative importance that production shares in comparison to transfer functions (of sediment and water) (Bedient *et al*, 1992; Linsley *et al*, 1975; Schumm *et al*, 1977). Defining this point is always a judgement call. In reality a diversely partitioned continuum of water and sediment runs from the ridgetops of Leading Creek down into the Ohio River, and beyond.

The hydrologic production and hydraulic transfer function aspects of the fluvial system may be viewed in consideration of some final depositional fate of sediment discharged to the Ohio River. Surface water run-off can be viewed as a functional element of the familiar continuous hydrologic cycle (Fetter, 1994; Linsley *et al*, 1975; Bedient *et al*, 1992), in balance with losses to deep groundwater, and the global climate (Bicknell *et al*, 1993; Donigian *et al*, 1995; Fetter, 1994; Freeze, *et al* 1979).

The conceptual fluvial model provided is helpful in underlining the concepts that:

- Hydrology is complex and ultimately determines the values of input variables used in most hydraulic equations (Bedient *et al*, 1992; Diplas, 1997; Donigian *et al*, 1995; Linsley *et al*, 1975; Yang, 1996)

- For a given time period, relief, geology, and climate, any observed disturbances to hydrological systems are likely initiated by changes to the land surface vegetation or land surface relief (Leopold *et al*, 1964; Schumm, 1977),

- Significant stress or instability manifested within the conveyance zones of either the drainage network or the channel-valley unit indicates that hillside features are the most logical candidates for identifying the original source of that stress. Exceptions would include effects of direct channelization and overt changes to relief like dams (Diplas, 1997; Lane, 1955).

On a closing point, the prophetic quote by Langbein is offered to any student of stream mechanics. His quote: *"Rivers are authors of their own geometry"* underscores the importance of respecting "dynamic stability" or "dynamic equilibrium" (Diplas 1997; Hack, 1960; Strahler 1957). Understanding this concept is requisite in describing morphological structure, its origin, and its tendencies (Diplas, 1992b; Leopold *et al*, 1964).

2.6 CLIMATE

Shown above in Table 2.5, climate is a fundamental forcing function of hydrology and hydraulics for a given time period, relief and geology (Schumm, 1977). A brief review of basic climatological principles is given here using the study watershed as an example (Bedient *et al*, 1992; Linsley *et al*, 1975; Leopold *et al*, 1964). This discussion provides an overview and understanding of various regional statistics and data availability. This will assist in showing how data can be used and made available to study ecology in a watershed context. The data would be procured and obtained for example in developing a hydrologic simulation (Allan, 1995; Arnold *et al*, 1995; Bedient *et al*, 1992; Donigian *et al*, 1995; Linsley *et al*, 1975; Waters *et al*, 1995).

The following summary on regional climate trends for the Leading Creek Watershed can be developed from information provided by the National Climatic Data Center (NCDC) in Asheville, North Carolina. NCDC serves as a repository and clearing-house source of climatic data and other weather related information for the U.S and its territories. To establish an accurate range of historical values for the Leading Creek Watershed, data was gathered from three nearby regional National Weather Service (NWS) stations including Huntington and Charleston in West Virginia, and Columbus, Ohio. This climatic data could also be supplemented by rain-gage stations operated by Southern Ohio Coal Company (SOCCO) at Mines No 2 and No. 31, as part of their compliance with National Pollutant Discharge

Elimination System (NPDES) permits. Other more limited data was also identified for potential future use, primarily temperature and precipitation information at nearby "local" NWS stations.

2.6.1 Weather Stations

Points of interest regarding historic annual and monthly climatic statistics are summarized below and in Tables 2.6, 2.7, 2.8, 2.9. Tables 2.6 and 2.9 list and classify weather stations in close proximity to Leading Creek. Summary statistical analysis in Tables 2.7 and 2.8 was based on National Climatic Data Center statistics covering in most cases the last 30 years (precipitation and temperature 1961 to 1990). In some cases more or fewer years are averaged for each regional station. Except for temperature and rain data, averages usually included data through the year 1995.

For the areas of interest, climatic statistics are derived from the best information available at this time for the parameters described. Many of the parameters in Tables 2.7 and 2.8 are used for input into more sophisticated hydrologic simulation model routines (Arnold *et al*, 1990, 1995; Bicknell *et al*, 1993; Linsley *et al*, 1975). Seasonal and monthly variation of input parameters is typically required for proper calibration of such models (Linsley *et al*, 1975), particularly those sensitive to land use changes with seasonal cycles, like farming. Time sequence data, for example periodic precipitation totals, is also usually needed in appreciably smaller time steps (e.g. hourly, 15 minute, 5 minute) to drive the more sophisticated simulation routines that attempt to determine run-off characteristics (Bicknell *et al*, 1993; Donigian *et al*, 1995).

The National Oceanic and Atmospheric Administration (NOAA) provides detailed descriptions of each parameter discussed in Tables 2.7 and 2.8. The U.S. Department of Agriculture Soil Conservation Service (SCS) provides useful summaries of climatological data for various counties across the country. The best local descriptions for this watershed were found in data summarized for Athens County, Ohio. Calculations for Leading Creek presented here were found to be similar in nature and magnitude to those presented by SCS soil surveys in previous reviews for earlier time frames where mentioned (Lucht *et al*, 1985). There may be perhaps modest increases noted in precipitation by moving the record starting point from 1951 to 1960. Based on similarities and gradation observed between weather station sites, data in the aforementioned tables should be fair for use in describing Leading Creek's most recent 30-year climatic conditions. Specific time sequence precipitation and other data collected at sites closer to Leading Creek is also summarized below.

Table 2.6 - National Weather Surface Regional Observation Stations

Station Name	Station Location	Distance to Leading Creek	Latitude	Longitude
Columbus, Ohio	International Airport	119 km	39 59 29	–82 52 51
Huntington, West Virginia	Tri-State Airport	89 km	38 22 54	-82 33 18
Charleston, West Virginia	Yeager Airport	95 km	38 22 46	-81 35 29

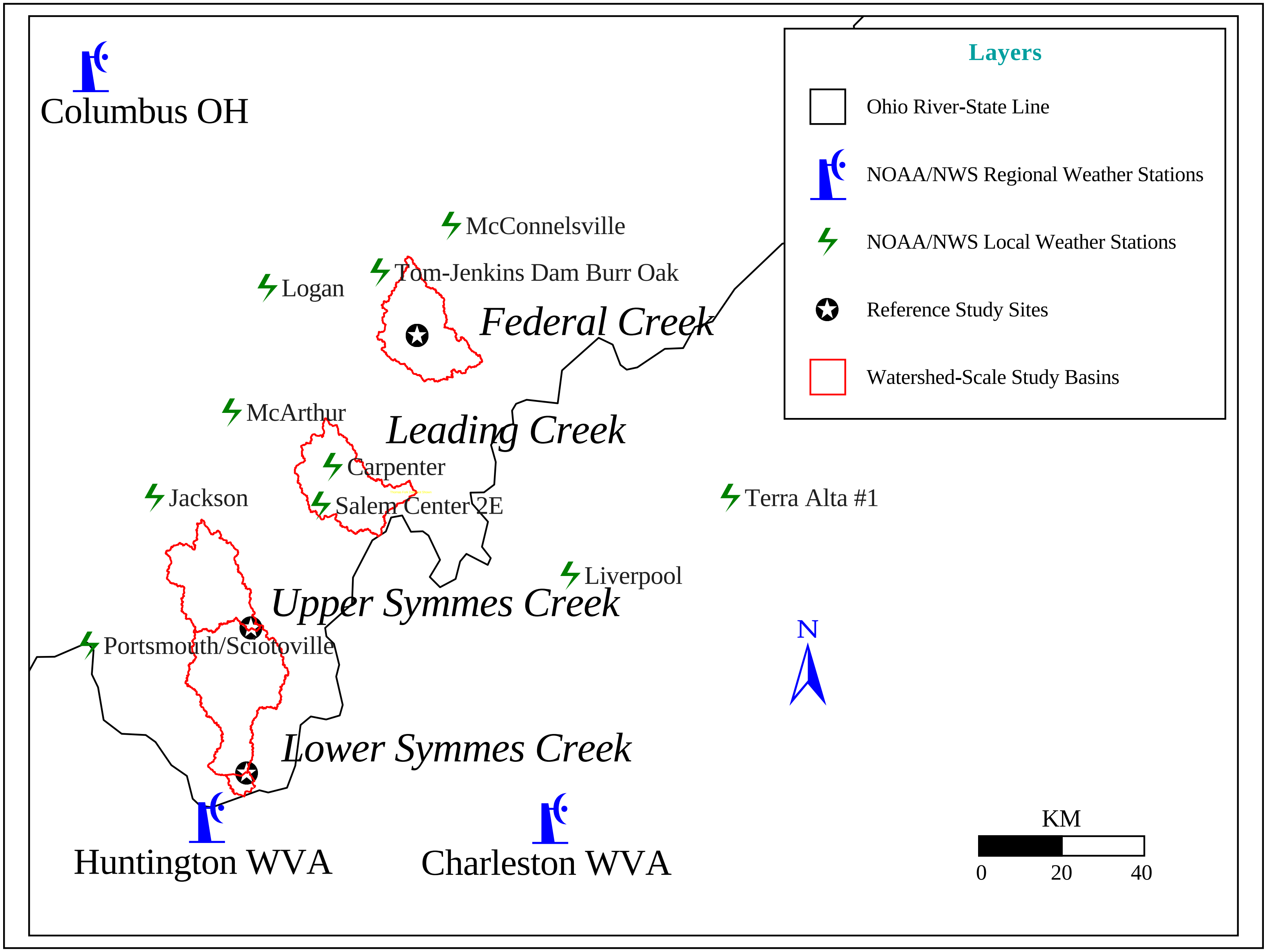


Figure 2.3 – Local and Regional National Weather Service Stations – Leading Creek and Reference Watershed Vicinity

Table 2.7 - Historical Annual Summary
Leading Creek Watershed - Regional Area NWS Weather Station Summary
All values below represent averages calculated on an annual basis
and include data from Columbus, OH, Huntington WVA, & Charleston WVA stations.

Average of the Three Regional Station's Annual Averages		
DataCode	**Description**	**Total**
NormPrecip	Normal Precipitation, Inches	40.7
AvgSnowfall	Average Snowfall (Including Ice Pellets)-Total in inches.	29.0
HighestTemp	Temperature - Highest of Record, Degrees F.	102.7
LowestTemp	Temperature - Lowest of Record, Degrees F.	-19.7
DaysBelow32	Mean Number of Days Minimum Temperature 32 F.or Lower	104.0
DaysPrecip>0.01"	Mean Number of Days with Precipitation 0.01 inch or More	142.7
HeatDegDays	Normal Heating Degree Days (July-June) Base 65 Degrees F.	5006.3
CoolDegDays	Normal Cooling Degree Days (Jan-Dec) Base 65 Degrees F.	944.3
AvgDailyTemp	Normal Daily Mean Temperature, Degrees F.	53.8
MaxDailyTemp	Normal Daily Maximum Temperature, Degrees F.	64.0
MinDailyTemp	Normal Daily Minimum Temperature, Degrees F.	43.5
AvgWindSpd	Wind - Average Speed (MPH)	7.0
%Sunshine	Sunshine - Average Percent of Possible	49.0
RelHumMorn	Average Relative Humidity - Morning(M)	82.0
RelHumAft	Average Relative Humidity - Afternoon(A)	57.7
MaxWind-SP	Wind - Maximum Speed (MPH)	51.3
MaxWind-DR	Wind (Maximum Speed) Direction	26.3
Cloudiness-CL	Cloudiness- Mean number of days (Clear)	66.7
Cloudiness-PC	Cloudiness- Mean number of days (Partly Cloudy)	104.3
Cloudiness-CD	Cloudiness- Mean number of days (Cloudy)	194.0

Table 2.8 - Historical Annual Summary
Leading Creek Watershed - Regional Area NWS Weather Station Summary
All values below represent averages calculated on an annual basis
and include data from Columbus, OH, Huntington WVA, & Charleston WVA.
Seasonal period values represent statitistics of monthly data given below.
USGS water year summer defined by April through September.

DataCode	Average Annual	Water Year		Calander Year Seasons			
		Winter	Summer	Winter (DJJ)	Spring(MAM)	Summer(JJA)	Fall(SON)
NormPrecip	40.7	18.1	22.6	8.6	10.9	12.2	9.0
AvgSnowfall	29.0	28.2	0.8	21.4	5.5	0.0	2.1
HighestTemp	102.7	89.3	102.0	78.7	93.3	102.0	99.7
LowestTemp	-19.7	-19.7	17.7	-19.7	-2.7	36.0	6.3
DaysBelow32	104.0	99.3	6.3	68.0	21.7	0.0	16.0
DaysPrecip>0.01"	142.7	75.3	68.0	40.0	40.3	33.0	30.0
HeatDegDays	5006.3	569.3	4437.0	21.0	967.3	2886.3	1131.7
CoolDegDays	944.3	20.3	924.0	0.0	82.7	724.0	137.7
AvgDailyTemp	53.8	40.6	66.9	32.9	53.6	72.6	55.9
MaxDailyTemp	64.0	50.1	77.8	41.6	64.7	83.1	66.4
MinDailyTemp	43.5	31.1	56.0	24.3	42.5	62.0	45.3
AvgWindSpd	7.0	7.8	6.2	8.1	7.9	5.6	6.4
%Sunshine	49.0	41.0	57.8	36.3	50.0	60.0	51.3
RelHumMorn	82.0	79.1	85.3	77.7	77.7	87.9	85.4
RelHumAft	57.7	60.1	55.4	64.0	52.6	57.3	57.1
MaxWind-SP	51.3	41.7	51.3	41.7	51.3	41.0	38.0
MaxWind-DR	26.3	25.4	25.8	24.0	25.7	27.2	25.6
Cloudiness-CL	66.7	32.0	35.7	12.7	16.3	16.7	22.0
Cloudiness-PC	104.3	39.0	64.7	17.3	24.7	37.3	24.3
Cloudiness-CD	194.0	111.0	82.7	60.0	51.0	38.0	44.7

Table 2.9 - Active Local and Regional Weather Observation Stations

NOAA Station	County/ State	Data Tape Index	Elev. (ft.)	15 Minute Rainfall Data?	Hourly Rainfall Data?	# Years	Lat	Lon
Local								
Tom-Jenkins Dam Burr Oak	Athens OH	8378	760	Since 1984	Yes	55	39 33	-82 04
McArthur	Vinton OH	5029	785	Since 1984	Yes	55	39 15	-82 29
Jackson NW	Jackson OH	4004	800	No	Yes	76	39 04	-82 42
Liverpool	Jackson WVA	5323	665	No	Yes	56	38 54	-81 32
Logan	Hocking OH	4672	722	No	Yes	53	39 31	-82 23
McConnels-ville - Lock 7	Morgan OH	5041	760	No	Yes	110	39 39	-81 52
Portsmouth/ Sciotoville	Scioto OH	6781	540	No	Yes	164	38 45	-82 53
Terra Alta #1	Ritchie WVA	8286	760	No	Yes	46	39 04	-81 05
Salem Center 2E	Meigs OH	N/A	741	No	No	Since 1963	39 03	-82 14
Carpenter	Meigs OH	N/A	821	No	No	Since 1978	39 08	-82 12
Regional								
Columbus	Franklin OH	1786	813	Assume Since 84'	Yes	55	39 59	-82 53
Huntington	Wayne WVA	4393	830	Assume Since 84'	Yes	34	38 23	-82 33
Charleston	Kanawha WVA	1570	1055	Assume Since 84'	Yes	46	38 23	-81 35

Compared to time sequence data collected inside the watershed, external time sequence rainfall data sets will likely have limited uses due to the lack of associated flow calibration data in Leading Creek for the same time periods. Previous to this study, Leading Creek was an ungaged watershed except for limited data collection conducted for a three-year period at Carpenter, Ohio. External data can, however, be acquired and compiled to generate a representative 30-year climatic cycle of the watershed, once a model has been compiled and appropriately calibrated with in-watershed data. For example, after initial calibration using internal time- sequence rain gage data sets and flow data sets (e.g. 5-minute time step), outside climatic data could then be able to be used to drive a longer-term simulation run-off model of Leading Creek (Bicknell *et al*, 1993; Linsley *et al*, 1975).

2.6.2 Weather Patterns of the Region For the Most Recent 30 to 50 Years

Weather patterns are described for stations in Figure 2.3 and Tables 2.7, 2.8 and 2.9:

- Average annual precipitation is approximately 40.7 inches. Of this 55.5% falls during the water summer or last half of the water year which includes the months of April through September. Precipitation is well distributed over all calendar seasons (8.6” winter, 10.9” spring, 12.2” summer, 9” fall).

- Annually, the number of days with precipitation > 0.01 inches is 143.

- Average seasonal snowfall is 29 inches, 74% falls during the winter; tends to be highly variable from year to year. Average normal daily mean temperature, annually, is 53.8 ^{o}F.

- Winter average temperature is 32.9 ^{o}F, average minimum daily temperature is 24.3 ^{o}F, with lowest on record –19.7 ^{o}F. Summer average temperature is 72.6 ^{o}F, average maximum daily temperature is 83.1 ^{o}F, with highest on record 102 ^{o}F.

- Prevailing maximum wind speed (max sustained for 1 minute) is from the west (Average 263 degrees from true north, ranging 240 to 272 degrees). Maximum wind speed is highest in spring, ~ 51.3 miles/hour.

- Highest at night, average annual relative humidity is 82% in the morning and 57.7% in the afternoon.

- Average annual and spring % of possible sunshine is 49 to 50%, and is 60% for the summer months.

- Prevailing average wind speed is highest in winter and spring, ~ 8 miles/hour, and annually 7 miles/hour. Athens County data analysis, period 1951 to 1978, indicated prevailing (assumed average) wind speed direction was from the northwest.

- Annually there are on average 66.7 clear (CL) days, 104.3 partly cloudy (PC) days, and 194 cloudy (CD) days.

- Thunderstorms may occur approximately 45 days of the year, most in summer. Rainfall is appreciably heavier on windward, west facing slopes than in valleys. Intermittent thaws typically preclude long-lasting (significant) snow cover in the region. (Lucht *et al*, 1985).

- For Athens County, which is directly north of the study site, from 1951 to 1978 annual precipitation was 38.6 inches and the average annual snowfall was 14 inches with greatest maximum snow depth of 5 inches at any one time. Maximum daily rainfall was 3.65” July 12, 1966 (Lucht *et al*, 1985).

2.6.3 Time Sequence Climatic Data Sets

Shown in Figure 2.3 along with regional weather stations, NOAA also maintains data collection at local (or primary) and secondary stations located closer to the watershed (e.g. Athens, Ohio), and inside the watershed (e.g. Carpenter, Ohio). Like the regional observation stations, detailed data can also be derived from smaller primary station operations described in Table 2.9. Compared to regional NWS stations, primary or local stations listed typically have fewer parameters available and longer time steps for data collection (e.g. daily or hourly instead of 15 minute rainfall data for example).

Stations of most interest for potential future study are found in Table 2.9. All have some form of daily information. Based on distance from site, the stations have limited utility though for time sequence hydrologic simulation, compared to in-basin gages available through monitoring conducted in this study and data collection associated with NPDES permits issued to active mining operations adjacent to and within the Leading Creek watershed. Sites in Table 2.9 could be useful for comparing weather patterns and checking accuracy and patterns of continuous in-basin stream stage gages. Hourly and 15 minute data needs to be purchased from NCDC. The NCDC data sets are continually updated. Costs in 1999 would be approximately $300 per 112 years of equivalent hourly station data, per tape loaded (e.g. Tape-3260 = 15 minute time step data, and Tape-3240 = hour time step data). The NCDC data retrieval system can be accessed relatively easily and quickly (1-828-271-4800 or via its Internet website).

Locational data in Tables 2.6 and 2.9 was taken from direct conversations with NCDC staff, Charleston NWS Station Staff, or NCDC 3240/3260 1996 Station Lists for Ohio and West Virginia. The in-watershed station listed above at Carpenter, Ohio (only daily information available) was located at a nearby location from 1978 to 1992 (Lat 39 12, Lon -82 17, elevation 879 ft.). Current climatic data collection at Carpenter is provided by a "citizen site-caller" Mr. Larry Montgomery (704-669-6065) located in Carpenter, Ohio who reports some daily rainfall data to the Charleston WVA Station. Currently, the NWS Charleston Station handles all weather forecasting for the area encompassing the Leading Creek Watershed.

2.6.4 Evapotranspiration

Evaporation rates for reservoirs and pan-coefficients for the regional area can be estimated from NWS data summarized by Linsley (Linsley *et al*, 1975). Solar radiation is the most important meteorological factor determining evaporation, followed by air temperature, vapor pressure, wind and possibly atmospheric pressure. Evaporation may vary with latitude, season, time of day, and sky condition (Linsley *et al*, 1975). A Weather Bureau Class-A pan coefficient for southeast Ohio may range 0.70 to 0.75. This pan type commonly used in hydrologic studies represents a surface exposure estimate of evaporation as opposed to use of floating or sunken pans.

Sunken pans tend to exhibit higher values than surface-pan derived estimates (Linsley *et al*, 1975). The pan coefficient represents an annualized ratio of lake-to-pan evaporation, and can be consistent from year to year (Linsley *et al*, 1975), despite variations in weather. In the Ohio River Basin area evaporation was estimated in the range of 32 to 36 inches per year, with an average for Leading Creek estimated at just above 34 inches/year or 86 cm/year (Linsley *et al*, 1975).

The approach of using pan rates in hydrologic simulation circumvents an otherwise relatively complicated approach to solving evaporation rates based on a separate or mixed combination of techniques involving a water-budget, energy-budget, and aerodynamic principles. The latter approaches rely on water-temperature data. As needed, the pan coefficients given can be applied to regional observations of actual pan-evaporation collected by NCDC and NWS for the time period of study.

2.7 GENERAL GEOLOGY - BEDROCK

The geologic setting for the study area is within the Interior Lowland topographic province. Geology is characterized by horizontally oriented sedimentary bedrock. Beds are characterized by northeast-southwest strike and average dip of 30 feet per mile towards the southeast (Lucht *et al*, 1985). Flat- lying sedimentary rocks seen in the Leading Creek Watershed are widely distributed, but topographic relief commonly intersects many rock types over short distances (USEPA, 1980). Other summary points of geologic interest and description specific to the Leading Creek Watershed area are provided below (Cardwell *et al*, 1968; Owens, 1967; Summerson, 1962; personal communications with R. Wilson, C.P.G, 1998). Cecil (1985) describes important paleoclimate controls on sedimentation and peat formation for Appalachia.

2.7.1 Geologic Units

Underlying geologic units include members of the Pennsylvanian age Conemaugh Group (Pc) and Allegheny Formation (Pa). Depositional environment varies between marine and non-marine conditions, and is distinguished by multiple coal sequences (Wilson, 1998). The worst acid mine drainage (AMD) discharges in Appalachia are preferentially associated with the marine coals, or with coals interbedded with marine strata (Caruccio *et al*, 1974; Younger, 1996). These strata will often have the highest sulfur contents, reflecting seawater composition.

Surface lithologies in Leading Creek vary between the Pennsylvanian age Monongahela Group (Pm) and the Pennsylvanian/Permian age Dunkard Group (Pd). The depositional environment was a non-marine Pennsylvanian – Permian basin also distinguished by multiple coal sequences, with likely minor amounts of limestone. Estimated time of sedimentation is between 280 and 330 million years ago, during the Paleozoic era at the end of the Carboniferous period. Deposition continued into the Permian epoch (280 million years) within an area known as the Dunkard Basin – a zone trending east and north of the study area. Dunkard Group rock out-crops may occur and would be found at the highest topographic exposures.

Dunkard Group – An estimated ≥135m thick non-marine sequence of sandstone, siltstone, shale and coal, subdivided in some locations into the Greene Formation, Washington Formation and Waynesburg Formation. Includes the Washington coal. Estimated age is between 265 and 285 million years.

Monongahela Group – An estimated 90m thick sequence of non-marine sandstone, siltstone, shale and coal subdivided in some locations into the Uniontown Formation and Pittsburgh Formation. Includes the Pittsburgh, Uniontown, Waynesburg, Sewickley, and Redstone coals. Estimated age is 290 million years.

Conemaugh Group – An estimated 160m thick assemblage of mostly non-marine shales, siltstone, sandstone and coal, with some limestone subdivided in some places into the Casselman Formation and Glenshaw Formations. Coal units include, Upper Freeport, Elk Lick, Bakerstown, and Mahoning seams. Estimated age is 300 million years.

Allegheny Formation – An estimated 100m thick assemblage of marine sandstone, siltstone, shale, limestone and multiple coal units (Freeport, Kittanning, Clarion, Princess). Estimated age is 330 million years.

Bedrock associated with abandoned surface and near-surface underground mining conducted above stream channel elevations in Leading Creek lies within the Monongahela Group. Within that group, the near surface Redstone coals were popularly mined in Leading Creek. The upper lithologies in Leading Creek's ASML mining district drop from massive sandstone cap rock through a 1 m shale layer lying above the Redstone #8A coal bed. Active deep long-wall underground mining in Leading Creek is conducted in the Clarion #4A coal bed.

2.7.2 Bedrock Classification and Acid-Sensitive Areas

Because of its similarity and applicability to understanding key mechanisms associated with acid mine drainage (AMD), the following notes on acid rain deposition that also induces stream acidification has been included here. Research on stream acidification due to AMD or acid rain indicates both sources lead to channel discharges with increased sulfuric acid (H_2SO_4) (Nordstrom, 1982; Skousen, 1996; Snoeyink *et al*, 1980; USEPA, 1980). Acidification may be defined as a decrease in alkalinity due to an equivalent increase of strong acid input. Alkalinity represents a system's net buffering capacity to neutralize input of strong acid. In natural buffered waters, total alkalinity is usually analytically defined as the sum of $[HCO_3^-] + 2[CO_3^{2-}] + [OH^-] – [H^+]$ expressed as molar equivalents per liter (Snoeyink *et al*, 1980; Evangelou, 1995).

Excerpts from EPA's study on stream buffering capacity can be indispensable in translating a brief background of AMD impacts in the eastern United States watersheds. The USEPA's acid rain study remains a timely biological reference covering adverse impacts of AMD induced stream acidification. The study also points out that regional geology plays a fundamental role in determining biotic integrity related to stream acidification (USEPA, 1980).

2.7.3 USEPA's Bedrock Classification and Stream Acidification Study

2.7.3.1 Study Objectives

USEPA recognized the benefit of a rock classification system to predict and better manage areas susceptible to acid deposition. The system developed was based on the buffering capacity of each rock type classified, and residing within a given area. Objectives of the sensitive areas mapping project were to evaluate the eastern United States by (USEPA, 1980). Objectives included: (1) Determining which areas were most vulnerable to adverse impacts from acid rain deposition, and (2) Reviewing biological consequences of freshwater acidification, and regional assessments of effects on fish, which can provide a basis for predicting such impacts.

At the time of establishing categorical effluent limitations for the coal mining industry, an USEPA survey of potential AMD impacts associated with coal washing plant operations and NPDES effluent discharges also recognized similar regional bedrock buffering concepts (USEPA, 1976a, 1976b; 40 CFR 434 – Code of Federal Regulations).

2.7.3.2 Study Results

Three major factors identified by USEPA as being important to an aquatic systems susceptibility to stream acidification were meteorology, pedology, and geology. Rock materials were classified into four broad types: Classes I, II, III, and IV, representing increasing buffering capacity ranging from none (Class I) to infinite capacity (Class IV). Associated impacts to biota for each class ranged from widespread impact to no impact. Supporting biotic response data was primarily qualitative. Conclusions of the USEPA study on stream acidification focused only on the aspect of geological control of sensitivity to acidification and indicated that (USEPA, 1980):

- Acidification of streams from atmospheric deposition (pH < 5.6 s.u.) with associated adverse impacts is occurring in all contiguous states from the Mississippi River to the Atlantic Ocean, from New England to Florida.

- Acidification as atmospheric deposition had effects that were similar to AMD, though AMD toxicology and chemistry were obviously complicated by the presence of high concentrations of heavy metals, chemical flocs, turbidity, and other components.

- Impact to aquatic ecosystems from acidic atmospheric deposition was largely based on the chemical characteristics of bedrock.

- Results suggested that regional bedrock geology exerted the strongest influence on aquatic organisms. Regionally, soil and vegetative types were of secondary importance to acid rain deposition (refer also to Richards *et al*, 1994b, 1996).

- Additional controls on acidification which may temper conclusions based on bedrock may included:

    - Hydrogeologic characteristics of terrain (soil thickness, permeability, porosity, etc.),
    - Types of soils (residual, glacial, aeolian, lacustrine, alluvial, etc.),
    - Mineralogy (e.g. quartz, coal, etc), and
    - Age of soil (typically mature/older soils offer more buffering capacity).

- Limestone or dolomite terrain yielded infinite buffering capacity.

- Because small amounts of limestone and dolostone materials can have an overwhelming influence on acidification potential, locally, soils in certain drainages could overwhelm regional bedrock influences.

- Type II Class included rock materials associated with sandstones, shales, and conglomerates with no free phase carbonates present. Although not directly classified in the earlier study, the Leading Creek Watershed is predominately underlain (~90%) by rocks from the Type II class, having medium to low buffering capacity.

- Based on non-AMD acid deposition rates only, predicted adverse impacts for Class II were limited to 1st and 2nd order streams and small lakes (scale: 1:125,000 to 1:500,000). Effects were normalized to a gradation of pH.

- The USEPA's study was completed using primarily maps at scale of 1:250,000 to 1:500,000 and larger for bedrock, and some smaller scales (e.g. 1:125,000) for study sites. In this scheme, 1st and 2nd order streams equate to roughly 3rd through 4th order streams at a scale 1:24000. By assumption 1st and 2nd order streams at a scale of 1:24,000 would be presumably impacted to an equivalent or greater degree (Aronoff, 1993; McCammon, 1998; Minshall *et al*, 1985; Vannote *et al*, 1980).

2.7.3.3 Study Summary

Only a sparse classification of bedrock in Ohio was performed in the USEPA study, shown in the referenced Figures II-4b,4d (USEPA, 1980). The report indicated only a 10-20% portion of Type II rock materials present in adjacent Mason County, WVA to the south, and in Ohio, Washington and Monroe Counties to the northwest of Athens, along the Ohio. In the coal belts of West Virginia and Kentucky rock classification runs 70-90% Class II rock. In all of the above designations mentioned, 0% Class I rock was also typically identified. Meigs, Gallia, and Athens Counties were not directly or generally classified, perhaps due to lack of soil surveys completed at the time of publication. Unpublished Ohio EPA sections collected in Leading Creek indicated a thin limestone in surface lithologies only at the northwest tip of the watershed.

The USEPA study found that drainage-scale geology dominated local aquatic response to acidification. The study clearly identified acidification, buffering, and dilution as key players in decreasing overall biodiversity. Literature generally recognizes two significant stages of acid generation associated with pyretic AMD succession (Evangelou, 1995; Nordstrom, 1982; Snoeyink *et al*, 1980). The study acknowledged metal toxicity co-occurs in AMD problems and other stream acidification scenarios. The USEPA acid deposition study indicated pH itself might not necessarily be a surrogate variable for metal toxicity when defining biotic response to AMD., but has direct effects. Mortality and overall depressed biological condition can originate simply from low pH and associated side-effect consequences of AMD acidification.

Both pH and heavy metal toxicity, individually, and in concert, are expected to adversely and measurably impact biotic production indices to the magnitudes that have been observed in AML/AMD areas in Leading Creek (Burton, 1992; Chapman *et al*, 1990, 1992; Cherry *et al*, 1995, 1996, 1997; Herricks *et al*, 1985). While not offering a clear pathway to separate effects of the two mechanisms (i.e. acidity and heavy metal toxicity), USEPA's acid deposition study together with water quality assay data (USEPA, 1996a) indicate both mechanisms can be equally and separately important in assigning AMD's adverse impacts to biota. So for example, the exact nature of geochemical interactions and acid-base equilibrium conditions need not be known exactly or even at all, where pH may perhaps still predict fairly well impacts to biota (Allan, 1995; Gallagher *et al*, 1999).

2.8 SOILS

To better understand the study watershed, a review of soil series expression and relief was completed in Leading Creek, moving from headwater areas to the Ohio River (Arnold *et al*, 1990; Bedient *et al*, 1992). This was a useful exercise to begin to assess soil variation in the watershed and to anticipate implications upon current land use/cover and management practices.

The soil data described here, including written and map information, was compiled from the USDA soil surveys available for each county hosting Leading Creek, (Feusner *et al*, 1989; Gillmore *et al*, 1991; Lucht *et al*, 1985). Specific aspects associated with mining disturbances of upland soils were investigated eventually through more intensive GIS and spatial analysis using this information and other sources of geodata. The following information is critical to understanding the basic landscape determinants that affect how AG and AML influence ecology.

2.8.1 Description of Historical Erosion Processes

Climate and denudational processes, and the presence of tectonic forces determined placement of soils in the current landscape (Leopold *et al*, 1964). More recent tectonic forces are themselves also often associated with alluvial terrace structures occurring in the channel-valley of Leading Creek. As seen in Table 2.5 the idealized fluvial model relates that climate itself is the most relevant factor determining landscape form for a given time period, initial relief, and geology. The distribution of precipitation temperature, and wind are the most important overall factors (Leopold *et al*, 1964).

Each soil layer is the result of a complex process of chemical and physical weathering. Soil is derived mainly from host bedrock and water. Erosion, a strictly gravitational process, is associated with the mechanisms of entrainment and deposition, which together with weathering form the cycle of landscape denudation (Leopold *et al*, 1964). The Leading Creek Watershed falls within a morphogenetic region described by its mean annual rainfall (40.7”) and temperature (53.8 ^{o}F). The morphogenesis relates regional climate to trends in denudation for that region. The natural site-specific morphogenetic traits in Leading Creek include, in order of importance: mechanical weathering, chemical weathering, running water, and mass movement (Leopold *et al*, 1964).

A few notes on erosion and sedimentation are added for perspective (Diplas, 1997):

- One third of the world’s sediment reaches the ocean via 3 rivers in China.

- A worldwide average estimate of natural denudation of the earth’s surface is about 1 inch every 1000 years. Current rates for the Mississippi area may be up to twice that much, areas of the Appalachian region may be less (Leopold *et al*, 1964).

- Farming areas with the richest soil in the world, located near the Gulf of Mexico, have lost about more than a meter of topsoil thus far where the associated delta is as big as the State of Virginia.

- ♦ Historians have noted that many times in history, civilizations have been observed to till the topsoil approximately 50-60 generations, and then disappear.

2.8.2 Leading Creek Soil Survey Data

Soils at the earth's surface are typically viewed in a vertical profile of soil horizons (Lucht *et al*, 1985). Soil surveys completed in the counties of Leading Creek reported soil profiles by surface layer, subsoil, substratum, and bedrock. Each layer has its own distinct set of properties (e.g. texture), with overall properties imparted to the entire profile based on representative, but limited field data (e.g. permeability, water capacity, etc.).

The top surface layer is also referred to as the plow layer or topsoil. The bottom most substratum imparts direct qualities of the parent bedrock material. The inter-layered subsoil represents the transitional continuum of soil, and may be broken down into several soil horizons if needed to describe this portion of the cross-section (Lucht *et al*, 1985).

Athens County (Lucht *et al*, 1985) had the most thorough survey available to date for the Leading Creek watershed area, with a single binding and published maps (Athens). Gallia County (Feusner *et al*, 1989) and Meigs County (Gillmore *et al*, 1991) surveys were more briefly described, but were still complete in their address of soil associations and units. Meigs County data was based on fair quality blueline prints made of tentatively approved soil survey mapping of the county. Obtained in 1996, the Meigs County maps were awaiting final publication.

2.8.2.1 Depositional Traits

Sedimentary rocks, describing the original depositional environment that formed the bedrock (Lucht *et al*, 1985) underlie Leading Creek. Origin of soils is also described by depositional trait. Due to the presence of host sedimentary rock materials in the Leading Creek Watershed, terms alluvium, colluvium, and residuum apply. These terms describe the depositional environment in which soils were laid and formed.

The term *"alluvium"* represents unconsolidated materials of sand, mud, and other sediments deposited by overflowing water. *"Colluvium"* implies a deposit of rock fragments and unconsolidated material that accumulates by gravity at the base of steep slopes (Lucht *et al*, 1985). *"Residuum"* is an unconsolidated, partly weathered mineral material accumulating over solid bedrock, formed from the underlying bedrock itself. Residuum, like its metamorphic equivalent saprolite, represents a zone of gradation between what defines bedrock and what defines the substratum. *"Loess"* indicates material transported and deposited by wind. *"Lacustrine"* is fine silt and clay material deposited on glacial lake bottoms (Lucht *et al*, 1985).

2.8.3 Major Landscape Disturbances and Effects

For identification of short-term instream sedimentation sources, it is helpful to realize that materials may often be derived from close to the flood plain, the floodplain itself, or from within the channel (Diplas, 1997; Dunne *et al*, 1978; Leopold *et al*, 1964). This conclusion does not necessarily apply to areas with significant upland disturbances including significant mass movements (e.g. landslides, surface contour mining or rock quarries).

As discussed above in the section on historical erosion, the morphogenetic basis in strip mined areas in Leading Creek often appear to be turned upside-down, where mass movement and wash-off have come to dominate the last 50-100 years. River control (e.g. via hydroelectric power impoundment or lock operations) can also induce significant bed elevation changes to both upstream and downstream portions of the stream (CVC, 1998; Lane, 1955; Leopold *et al*, 1964).

Believed to have occurred with surface mining in Leading Creek, disturbances on steep to severely steep terrain can lead to tremendous increases in erosion on those hillsides. Accompanying washoff can swallow smaller nearby drainages of the network. In channel-valley areas downstream of mining, sediment flow increases ranging 100% to 1000% of normal bedload may be possible. Changes up to 100%-300% were noted even at points far downstream of placer gold mining (Leopold *et al*, 1964). The latter upper estimate obviously depends on the streams current configuration and capacity to carry added loads of suspended sediment delivered from the landscape. The point made is that mining can cause radical increases in sediment load.

Severe and significant adverse impacts to the natural channel-valley regime may reach 10-100 miles upstream and downstream of the actual disturbed area (Lane, 1955; Leopold *et al*, 1964). Time frames of a few years to 50 years to perhaps 100 years and longer, with an average of perhaps 15 to 30 years, may typify the rise and fall of severe sediment impact from man's activities (Diplas, 1997; Leopold *et al*, 1964). Time-duration appears consistent with knowledge that perhaps 10 to 25 or 30 significant bankfull storms might be needed in the interim conceivably to clear an inundated channel. There are leading and lagging edges of an effective AML-sediment storm manifested in a sediment hydrograph as sediment depth increases over decades of time and then subsequently decreases (Leopold *et al*, 1964).

Thus, AML sedimentation impact scenarios may be ideally thought of as a sediment-graph, similar to a storm hydrograph, but with a much greater time of rise and fall. In Leading Creek, the rising sediment stage would monitor sands and fines as opposed to water head normally measured in a hydrograph. These types of data records have been previously modeled by other researchers using mean low data taken at larger river towns over decades of time. The relative net bed depth datum gradually will creep up and then back down again, due to sedimentation and scour (Lane, 1955; Leopold *et al*, 1964).

Capping the discussion on the overt transformation of landscapes by humans, the following excerpt from Luna B. Leopold's classical text on fluvial geomorphology provides an important insight as to how humans interact with the landscape (Leopold *et al*, 1964):

> *" … as a rule geomorphic effects produced by man are the same as those produced without him. Usually man simply changes the magnitude of certain variables in the system. These in turn produce responses, perhaps only acceleration or deceleration, in the fundamental geomorphic processes. The appropriate principles are not abrogated."*

2.8.4 Soil Series Review and Watershed Analysis

The initial soil series reviews discussed here provided a particularly useful differentiation between agricultural practices and abandoned mined lands (AML) in the watershed. The approach taken to review county level soil surveys looked at occurrence, domination, and trends within single subwatersheds and across multiple subwatersheds, for each significantly occurring soil series. This was helpful in initially documenting the variation and presence of these land-uses and to further understand their associated geologic settings.

The soil map review generalized a more detailed stream bed slope analysis conducted as part of the study. It did so by primarily concentrating on 1st cataloging mainstem and major tributary-stem channel-valley units, and then assessing higher relief valley terraces as (frontslope) transitional zones to midslope and crestslope uplands. The review then assessed overall upland features on a tributary by tributary basis (the hillslope/drainage network). The basic morphological separation used was for grossly characterizing each area of the watershed:

- Channel/Valley (or frequently flooded plain),
- Frontslopes and infrequently flooded terraces transitioning to upland, and
- Hillslope-Uplands (midslope, crestslope, crest)

Each level of review appeared to accurately cross-reference soil series trends to known trends in land–use/cover profiles in each subshed studied. This effort served as some confirmation of the applicability and accuracy of the independent data sources of mining land-use information gleaned from geodata (e.g. Landsat5 and USGS 7.5-minute quadrangles). Detailed notes from the tributary by tributary soil survey reviews conducted were not repeated here but were documented in an initial risk assessment of Leading Creek (Cherry *et al*, 1999).

The informal review approach attempted to classify the three broad categories given, and identified dominant soil series occurring in each category, for each major tributary area of the watershed. With respect to the idealized fluvial model, this was done on more individual detailed subwatershed-scale and subshed-scales, reviewing basic soil series expression and transition to and from upland and the streambed. Channel areas of the smaller subwatersheds and subsheds were also looked at in this way. In the 3-prong conceptual model of Leading Creek discussed earlier, smaller channels would be otherwise assigned to the hillslope/drainage network morphological unit of modeled subwatersheds.

2.8.5 Digital Soils Data

Digital soil data was unavailable in Meigs County, comprising 96% of the watershed. Analysis of actual soil data was limited to the informal qualitative summary provided. Soil unit analysis helped to better define needs and potential that digital data could fill, particularly in establishing a more impartial, quantitative analysis of drainage-scale soil properties. A refined database incorporating digitized county level soil surveys would be needed to advance further study of Leading Creek's soil profiles, and associated sedimentation issues. Associated soil attribute properties, aereal extent, and vectorized digital basin-wide elevation (hypsography) data were not currently available in this project's GIS capability. Together, such digital data would magnify the ability to further resolve water and sediment interactions on a finer scale, greatly facilitating hydrologic simulation (Bicknell *et al*, 1993; Linsley *et al*, 1975; USEPA, 1981).

2.8.6 Morphological Units, Major Soil Associations, and Important Soil Series

Excluding disturbed mined lands, dominant soil series encountered across Leading Creek's hillside and channel-valley morphologies are silt loam texture classes (Gillmore *et al*, 1991), with minor exceptions. The silt loams encountered would be expected to have a silt content in the range of 60-85%.

The disturbed, unreclaimed AML lands are typified by sandy and sandy loam soils, comprised of a mixture of original silt loam series, capstone, and portions of underlying bedrock. Underlying bedrock in most AML areas is primarily sandstones and shales. The disturbed AML sands and sandy loams are expected to range 65-100% sands (diameter < 2mm), with a median diameter (D_{50}) in the medium to fine range (0.125-0.5mm). AML soils have higher percentages of gravel range particles (>2mm) compared to background silt loams (Lucht *et al*, 1985).

2.8.6.1 Channel-Valley

Soil associations in the channel, floodplain, and low lying terraces along Leading Creek and major tributaries (channel-valley morphological unit) are comprised mostly of deep well-drained Nolin (No) and Chagrin (Cg) silt loams, and to a lesser degree Newark (Nk) silt loams. These units are for the most part frequently flooded and represent, with varying degree, "prime farmland" in the watershed as defined by Department of Interior (Gillmore *et al*, 1991). The alluvial Chagrin (Cg) and Nolin (No) soils were formed in recent alluvium and represent the youngest soils, distinctly showing little or no differentiation of horizons (Lucht *et al*, 1985).

The Nolin (No) has a seasonal high water table of 48 to 72 inches (depth to groundwater from the land surface). For the Chagrin (Cg) soil series it is 36 to 72 inches, and for the more poorly drained Newark soil, 6 to 18 inches. The Nolin-Chagrin Association in general consists of soils on flood plains along major streams that are subject to rare or frequent flooding. These flood plains occur as two benches, one at a higher elevation than the other does. Valley areas along flood plain courses on mainstem and larger tributaries are approximately 0.4 to 0.8 km wide (Lucht *et al*, 1985).

2.8.6.2 Hillside - Uplands

Excluding disturbed mined lands, dominant soil series encountered across Leading Creek's uplands are also mostly silt loam texture classes. The Upshur-Gilpin (Ug) Silt Loam complex in all but the central western and northwestern portions of the watershed dominates uplands. In upper hillside areas of the western portions noted, the Omulga (Om) series, Westmoreland-Guernsey (Wh) complex and Gilpin-Rarden (Gk) complex are encountered.

2.8.6.3 Mined Land Impacts

AML can have an appreciable disturbance with three typical signatures downstream (Bigham *et al*, 1996; Boult *et al*, 1984; Curtis, 1973, 1979; Daniels *et al*, 1995; Dickens *et al*, 1985, 1989; Gray, 1996; Nordstrom, 1982; Stark *et al*, 1994; USDA, 1985, 1998):

- Acid mine drainage (AMD) in the near-field,
- Deposition of sands/fines and precipitation of fine floc silts in the near and mid-fields,
- Leading to coated/colored sands and silts/clay in the far-field, e.g. far downstream.

A host of colored flocs and coatings can also be observed close to AMD. The colored sands that typify Leading Creek in the mining belt area near to the creek mouth were likely hematite (sand with an iron oxyhydroxide coating) though this was not confirmed through laboratory analysis (Bigham *et al*, 1996; Diz, 1997; Nordstrom, 1982). Summarizing an important point, sedimentation due to AML appeared more ubiquitous and reached farther in extent in its impact upon biota, compared to its chemical cohort, acid mine drainage (AMD).

In Leading Creek, colored flocs and coatings appear to dissipate as well in distance from the given AML/AMD site. Strip mine footprint profiles are noticeably concentrated in relief on steep terrain ranging from 780 feet AMSL to 660 feet AMSL. Recall that a relatively gently sloped interbedded structure governs bedrock character in the watershed (Lucht *et al*, 1985).

2.8.6.3.1 Disturbed (Mined) Land Soil Series

For AML, soil series in Leading Creek include the Pinegrove (Pn) Sandy Loam, Pinegrove (Pu) Silty Clay Loam, and to a lesser degree the Kyger (Ky) Sandy Loam. The two sandy loam soil units represent a mixture of natural materials and materials disturbed from mining activities, by and large, not yet reclaimed (Gillmore *et al*, 1991). They are the likely source of the continued sand deposition found inundating streambeds (Gillmore *et al*, 1991), along with materials previously eroded but not yet transported to the Ohio River.

The Pinegrove (Pu) Silty Clay Loam characterizes reclaimed mined land in Meigs County. Prior to this reclaimed distinction, unreclaimed soils were likely of the sandy loam class. Pinegrove (Pu) Silty Clay Loam areas would be considered reclaimed historical source areas of washed-off sediment. The abated sources though may still be responsible for residual storage of sand/aggregate on hillslope areas just below unreclaimed mining areas and in stream channels downstream. Reclamation is a loosely applied concept with a wide range of interpretation covering revegetation and soil amendment through and unsuccessful tree plantings.

The soil series Mine Dumps (Dp) represents for the most part obvious spoil piles of mining residuals in typically near level to strongly sloping areas in valleys and at the base of hillsides near coal mines, processing areas, and loading areas (Gillmore *et al*, 1991). They also include active AEP/SOCCO (underground) mine surface operations areas in Parker Run and Ogden Run. The clastic materials are comprised of coal fragments, roof shales, underclay, and rock fragments (Gillmore *et al*, 1991) and are presumed to be associated more with underground mining operations in the basin, laid nearby and around mine entrances.

The coarser sandy materials associated with the Pinegrove, Kyger, and Mine Dump soil series are not a distinction of natural formation/deposition, but directly relate to results of mining operations conducted in these areas. Mining was conducted prior to the soil surveys that disturbed the natural soil regimes and bedrock lithologies (Gillmore *et al*, 1991).

The silty clay loam unit identified is a direct attribute of reclaimed AML land, with an apparently more formidable topsoil cover ranging about 6”-12” in depth. Athens County has a unique set of its own mined land soil series descriptions: the reclaimed Fairpoint (Fa) series and unreclaimed Fairpoint (Fb), Bethesda (Bo) and Barkcamp (Ba) series. None of the unreclaimed Athens County soil series occur within the watershed boundary aside from the limestone quarry operations designated by Fairpoint soils. Located in one of the more northwestern headwater areas located in Athens County, the closed quarry is covered by the Fairpoint (Fa) soil series.

2.8.6.3.2 Differences in Pinegrove Soil Series

The Gallia County soil survey (Feusner *et al*, 1989) indicates slight differences in texture between the composition of unreclaimed Pinegrove (Pn) materials cataloged for Meigs County (Gillmore *et al*, 1991). This occurs in drainages that straddle both counties. On the most southern drainages of the watershed, the Gallia County Pinegrove (Pn) Sand transitions to the Meigs County Pinegrove (Pn) Coarse Sandy Loam. The Gallia soil designation indicates yellowish brown sand 0 to 8” in depth, underlain by up to 52” of loamy coarse sand (Feusner *et al*, 1989). This indicates the implicit variability found in county soil surveys.

The Meigs designation delineated a surface layer of brown very friable coarse sandy loam 0 to 2” in depth, with up to 78” of underlying substratum composed of a yellowish brown and strong brown, very friable, channery loamy coarse sand (e.g. layered with thin flat fragments of parent material). The Meigs soil series more closely resembles the Gallia Pinegrove (Pg) which was used for designating mined areas on more gently sloping land. The differences noted are, for purposes of this study, only subtle. The two series both represent highly erodible sources of large amounts of sandy materials which continue to make their way to the valley floor and into the network of stream channels.

The differences in soil profile and texture assigned above in the case of Pinegrove series are attributed to the general limitations inherent in the survey methodologies. The surveys are based on very limited soil sampling of major soil units in each county. Differences noted between Gallia and Meigs Counties could also easily be noted within each respective county. Actual soil profiles would be assumed variable across many spoil areas given the disturbed nature of these lands (Daniels *et al*, 1995). The study discussed the Pinegrove (Pn) series designated by Meigs County due to its greater aereal extent within the watershed (96%).

2.8.7 Distinct Mainstem Soil Transitions

2.8.7.1 Soil Associations

The following USDA soil associations, based on geographic regional-scales (scale 1:100,000-250,000), were identified as having significant potential importance to the hydroecology of the Leading Creek Watershed. The soil associations have been organized into their occurrence over hillside and channel-valley fluvial components defined above.

Soil associations given represent gross trends across subwatersheds, primarily classifying the predominant channel-valley soil association, and dominant upland association, to the extent practical at the given map scale. Depending on drainage geomorphic scale (Leopold *et al*, 1964), a given drainage may have any number of other soil series associated with it (Lucht *et al*, 1985). In some cases a drainage may exclude the distinct soil series that the regional association is identified with, and for field-scale areas can exclude the specific soil series mapped in that area.

2.8.7.2 Mainstem Transition Areas

The following soil descriptions give a generalized breakdown of the watershed, necessarily combining some channel and hillside qualities. Soil associations primarily describe upland areas while spatial groupings represented (e.g. mainstem channel reach associations 1, 2, and 3 given directly below) are based on major channel-valley soil transition points (e.g. on the mainstem of Leading Creek). The areas grouped below do not coincide exactly with the mainstem and tributary-stem trunk lines identified previously in the conceptual fluvial model for Leading Creek. The three mainstem and primary tributary-stem reaches noted are actually lumped together, and are represented by the last area group given below (e.g. mainstem channel reach 3). Three lines based on valley slope separated the steepest portion, the transition (toe) zone associated with some colluvium materials, and the main alluvial valley of the watershed.

1. **Omulga-Licking Association:** ***(Mainstem - Albany to Sisson Run)*** - Direct remnants of the Teays River network, pre-glacial alluvium and lacustrine sediments dominate, and cover certain western watershed headwaters in Sharps Run, Sisson Run, Ogden Run, and Dexter Run. These are the prime farmland areas based on favorable Omulga and Licking silt-loam soil texture classes found on the remnant high relief flood plain terraces. Characterized by deep, gently sloping to moderately steep, moderately well drained soils. This association has sporadic expression in upland terraces occurring further down the mainstem towards the Ohio River. Other upland areas including those directly along the mainstem and in extreme western-central areas are Upshur-Gilpin (Ug) series.

   The uppermost undisturbed mainstem channel areas found in Athens County are primarily of the Newark (Nk) series. Otherwise the Nolin (No) series dominates channel and floodplain relief well into Meigs County, down to the Sisson Run area.

2. **Gilpin-Rarden and Upshur-Gilpin Associations:** ***(Mainstem - Sisson Run to Malloons Run)*** **–** formed in residuum and colluvium from shale, siltstone, and sandstone. Excluding the Parker Run and Malloons Run areas, western uplands, which also drain some pre-glacial terraces described above, reside primarily in the Gilpin (Gh) and Rarden (Ra) soils formed in residuum. Southwestern subwatersheds Parker Run and Malloons Run, mainstem areas, and eastern subwatersheds follow the Upshur-Gilpin (Ug) complex formed in residuum and some colluvium. Gilpin soils are mostly weathered from underlying siltstone and sandstone bedrock, and Upshur soils are weathered from a red shale bedrock. More distinctive in this area along the mainstem, expression of the

preglacial Omulga terraces occurs with some silty, older alluvium terraces of the Taggart (Ta) and Gallipolis (Gb) Silt Loam series.

The mainstem channel relief in this area expresses a mixture of both Nolin (No) soils transitioning off and on with Chagrin (Cg) soils, and some Newark (Nk) soils.

3. **Upshur-Gilpin-Pinegrove Association:** ***(Mainstem – Malloons Run to Ohio River)*** **–** upland soils in this area are dominated by Upshur-Gilpin (Ug) series, where AML lands are now designated by Pinegrove (Pu and Pn) and some Kyger (Ky) series. Even more distinctive in this area of the watershed seen upstream on the mainstem, expression of the preglacial Omulga (Om) terraces increases. Along with the Omulga (Om) again the older "dirty silt" alluvium terraces of the Taggart (Ta) and Gallipolis (Gb) Silt Loam series are more widespread. The latter Taggart (Ta) and Gallipolis (Gb) terraces are also found similarly in the lower relief areas along Little Leading Creek and Thomas Fork tributary-stems.

   Formed in more recent alluvium and lacustrine sediments, the primary watershed channel-valley floor is dominated by Chagrin (Cg) soils with some Nolin (No) soils. In the lower mainstem valley areas, a transitioning to more varied mixtures of terrace structure and soils occurs. Below the confluence of Little Leading Creek, mainstem terraces show little trace of the preglacial character of the northwest portions of the watershed.

Figure 2.4 delineates the energy slope profile of the mainstem based on USGS 7.5-minute quadrangles and is helpful to compare results below. Figure 2.4 was developed from USGS 7.5-minute quadrangle land elevation contour lines (6.1m) crossing the mainstem, estimating rivermile at those sites. Added hydrologic-hydraulic background analysis is found in appendices. Individual undisturbed series, disturbed series, and soil complexes encountered in Leading Creek are shown in Tables 2.10, 2.11, and 2.12 and are based on map data at a scale of 1:15840 (Feusner *et al*, 1989; Gillmore *et al*, 1991; Lucht *et al*, 1985).

2.8.8 Watershed Soil Series Encountered

The soil series given in Tables 2.10, 2.11, and 2.12 identify soils with significant potential importance to the hydroecology of the Leading Creek. They are taken primarily from (Gillmore *et al*, 1991). Soil series ID systems change from County to County, where Meigs County descriptions were used if available. For simple description purposes, surface land slope is not delineated here. The Westmoreland soil series given below only occurred in Leading Creek and Athens County as a complex. The associated soil series (Wl) was assumed.

Soil properties given are derived from the actual complex, assuming data for the most moderate slope. Another example of the localized nature of the applicability of survey information, the Vandalia (Va) surface layer indicated a silt loam in Meigs County and a silty clay loam in Gallia and Athens County (Feusner *et al*, 1989; Gillmore *et al*, 1991).

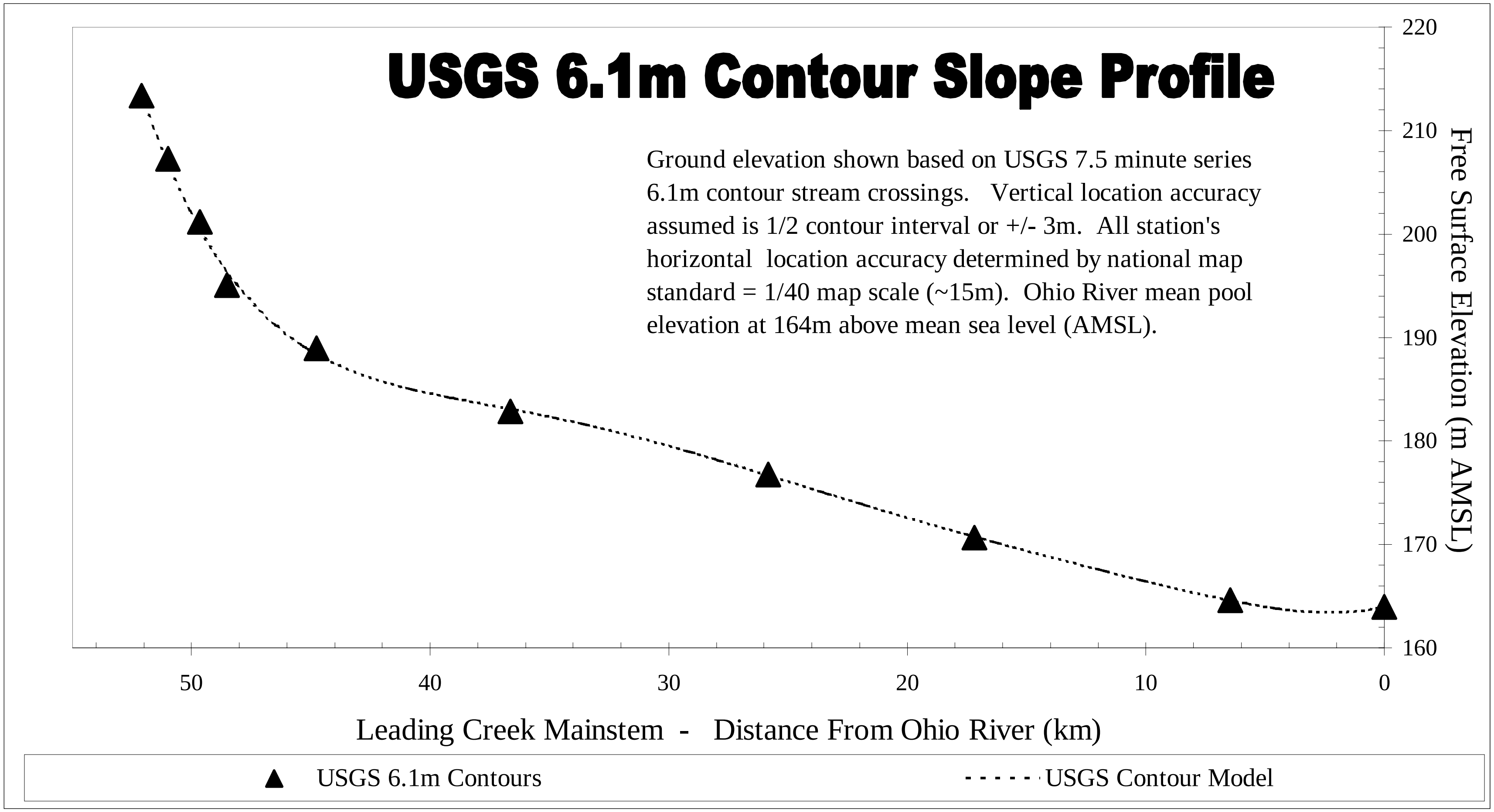


Figure 2.4 - Leading Creek Mainstem Energy Slope Summary

Table 2.10 - Undisturbed Soil Series Encountered In Leading Creek

Soil Series Name	Series Symbol	USDA Texture	Topography Occupied	Depositional Trait and Underlying Bedrock
Aaron	**Aa**	Silt Loam	Strongly sloping topography on ridgetops on uplands, including involved Aaron-Gilpin (Ag) and Aaron-Upshur (Au) complexes.	Residuum; weathered from shale and siltstone
Brookside	**Br**	Silt Loam	Frontslopes and benches below very steep hillsides.	Colluvium; weathered from shale and siltstone (likely)
Chagrin	**Cg**	Silt Loam	Nearly level topography on stream floodplains	Alluvium; Formed in deposits of recent alluvium
Elkinsville	**Ek**	Silt Loam	Nearly level topography on low terraces near small streams.	Alluvium; Formed in deposits of silty alluvium overlying loamy alluvium.
Gallipolis	**Gb**	Silt Loam	Nearly level and gently sloping topography on terraces along the Ohio River and smaller streams.	Alluvium; Formed in deposits of silty old alluvium.
Gilpin	**Gh**	Silt Loam	The Upshur-Gilpin (Ug) complex occupies strongly sloping to steep topography on ridgetops and side slopes of uplands, within the complex the Gilpin occupies steeper terrain; the Gilpin (Gh) occupies similar but gently to strongly sloping terrain; Gilpin-Rarden (Gk) complex and Guernsey-Gilpin (Gw) occupy similar topography.	Residuum; weathered from underlying interbedded siltstone, sandstone, and shale.

Table 2.10 (Continued)
Undisturbed Soil Series Encountered In Leading Creek **Page 2 of 3**

Guernsey	**Gs**	Silt Loam	The Guernsey-Gilpin (Gw) complex, typically occupies benches and other less sloping positions.	Colluvium & Residuum; derived from underlying siltstone and shale.
Keene	**Ke**	Silt Loam	Gently to strongly sloping topography on ridgetops on uplands.	Loess & Residuum
Licking	**Lk**	Silt Loam	Nearly level and gently sloping to moderately steep topography on terraces along major streams and on slope breaks on terraces.	Loess & Lacustrine; formed in thin layer of silty wind deposits and underlying clayey sediments.
Melvin	**Mh**	Silt Loam	Nearly level topography on stream floodplains.	Recent Alluvium.
Moshannon	**Mo**	Silt Loam	Nearly level topography on stream floodplains.	Recent Alluvium.
Newark	**Nk**	Silt Loam	Nearly level topography on stream floodplains.	Recent Silty (loamy) Alluvium.
Nolin	**No**	Silt Loam	Nearly level topography on stream floodplains.	Recent (Silty) Alluvium
Omulga	**Om**	Silt Loam	Gently sloping to strongly sloping topography in wide preglacial valleys.	Loess, Old Alluvium, and Lacustrine;
Orrville	**Or**	Silt Loam	Nearly level topography on stream floodplains.	Recent Loamy Alluvium.
Rarden	**Ra**	Silt Loam	Strongly sloping topography on ridgetops on uplands.	Residuum; formed from interbedded shale and siltstone.
Richland	**Rc**	Silt Loam	Gently sloping topography on alluvial fans	Colluvium; formed from interbedded shale and siltstone.
Steinsburg	**St**	Fine Sandy Loam	Steep and very steep topography of side slopes and shoulder slopes on uplands.	Residuum; formed in material weathered from weakly cemented sandstone.

Table 2.10 (Continued)
Undisturbed Soil Series Encountered In Leading Creek **Page 3 of 3**

Taggart	**Ta**	Silt Loam	Nearly level topography on terraces along major streams and the Ohio River.	Old (Acid) Silty Alluvium;
Upshur	**Ub**	Silt Loam	The Upshur-Gilpin (Ug) complex occupies strongly sloping to steep topography on ridgetops and side slopes of uplands, within the complex the Upshur occupies more gentle terrain; the Upshur (Ub) occupies similar but gently to strongly sloping terrain.	Residuum; weathered mainly from underlying red clay shale and siltstone.
Vandalia	**Va**	Silt Loam	Strongly sloping to moderately steep topography on frontslopes at the base of strongly sloping to steep hillsides.	Colluvium; formed in materials derived from shale, and lesser amounts of siltstone and sandstone.
Vincent	**Vn**	Silty Clay Loam	Gently sloping to strongly sloping topography in wide preglacial valleys.	Loess & Lacustrine; formed in thin layer of silty wind deposits and underlying clayey sediments.
Wellston	**Wd**	Silt Loam	Nearly level and gently sloping topography on broad ridgetops and saddles on uplands.	Loess & Residuum; formed in thin layer of silty wind deposits and underlying material weathered from siltstone, sandstone, and shale.
Westmoreland	**Wl**	Silt Loam	Westmoreland-Guernsey (Wh) complex occurs on ridgetops upper parts of side slopes, and on benches; complex Westmoreland-Upshur (Wm) complex occupies ridgetops, rounded knolls, hillsides, and along drainageways on steeper terrain.	Residuum; derived from underlying siltstone and sandstone.

Table 2. 11 - Disturbed Soil Series Encountered In Leading Creek

Soil Series Name	Series Symbol	Texture Class	Reclamation Trait
Mine Dumps	**Dp**	Aggregate - Variable	Variable, mostly unreclaimed lands, some reclaimed areas.
Fairpoint	**Fa**	Silt Loam	Reclaimed Land
Fairpoint	**Fb**	Shaly Clay Loam	Unreclaimed Land
Kyger	**Ky**	Sandy Loam	Unreclaimed Land ; Flooded for longer duration, brief ponding may occur
Pinegrove	**Pn**	Coarse Sandy Loam	Unreclaimed Land
Pinegrove	**Pu**	Silty Clay	Reclaimed Land

Table 2.12 - Soil Complexes Encountered In Leading Creek

Soil Complex Name	Symbol	Primary Soil Series Name	Secondary Soil Series Name
Aaron-Gilpin	**Ag**	Aaron (Aa) – 45%	Gilpin (Gh) – 35%
Aaron-Upshur	**Au**	Aaron (Aa) – 50%	Upshur (Ub) – 25%
Gilpin-Rarden	**Gk**	Gilpin (Gh) – 55%	Rarden (Ra) – 25%
Guernsey-Gilpin	**Gw**	Guernsey (Gs) – 45%	Gilpin (Gh) – 35%
Upshur-Gilpin	**Ug**	Upshur (Ub) – 50%	Gilpin (Gh) – 30%
Westmoreland-Guernsey	**Wh**	Westmoreland (We) – 40%	Guernsey (Gs) – 35%
Westmoreland-Upshur	**Wm**	Westmoreland (We) – 50%	Upshur (Wm) – 30%

Soil complexes are two or more commingled soil series that can't be practically further separated at the given map scale. Complexes are comprised typically of two soil series and miscellaneous material. In the complex, the series are still tracked individually, but may have subtly different properties than the individually occurring soil series.

2.8.9 Watershed Soil Properties

2.8.9.1 Soil Texture

To provide a sense of the range of permeability of parent rock materials and associated unconsolidated materials, the following, generalized summary on intrinsic properties is first given (porosity = $\text{Volume}_{\text{Voids}}/\text{Volume}_{\text{Sample}}$). Acidification buffering potential has also been expanded with estimates for generic unconsolidated materials (USEPA, 1980).

The texture of soil relates to its combined combination of clay, silt, and sand, and the associated textural qualities imparted (USGS Paper 1662-D, 1967; referenced by Lucht *et al*, 1985). Particle size statistics like a D_{65} (65% percent of material passing) and texture describes different lumped characteristics of the soil. They are only synonymous with size at soil separations shown. In the order of increasing fine particle content, with range of particle diameter shown, textures of soil are defined by (Lucht *et al*, 1985):

- **Sand** **(<2 mm)**
- Loamy Sand
- Sandy Loam
- Loam
- Silt Loam
- **Silt** **(< 0.074 mm)**
- Sandy Clay Loam
- Clay loam
- Silty Clay Loam
- Sandy Clay
- Silty Clay
- **Clay** **(<0.004 mm)**

The sand, loamy sand, and sandy loam classes can also be further described by “coarse”, “fine”, and “very fine” (Lucht *et al*, 1985). An example of this is the Pinegrove Coarse Sandy Loam Soil Series – (Pn). Particle size diameter, sieve #, and descriptions used in this study are based on standard engineering classification (Vanoni, 1977; Yang, 1996).

2.8.9.2 Particle Size and Other Properties

No specific chemical and physical sampling data for Meigs County soils were readily available. Data below is derived from the adjacent Gallia County database where possible, and from the Athens County database for certain series. Useful soil and bedrock properties are summarized in Tables 2.13, 2.14, 2.15, 2.16, and 2.17. Specific values for Leading Creek soils encountered are also featured in Table 2.18 - Particle Size Data, Table 2.19 – Hydrologic Properties, and Table 2.20 – Miscellaneous Properties.

Table 2.13 - Porosity of Parent Rock and Unconsolidated Materials
(Portions adopted from Fetter, 1994; Leopold *et al*, 1964; USEPA, 1980)

Rock	Acidification Buffer Class	Relative Permeability	Porosity
Granite	I – None	1	1%
Shale	II – Low-Medium	5	18%
Siltstone	II – Low-Medium	50	18%
Sandstone	II – Low-Medium	500	18%
Limestone	IV – Infinite	30	10%
Unconsolidated			
Clay	Medium	0.0001- 0.01	45%
Silt	Low-Medium	0.01-100	40%
Sand	Low-Medium	1000	30%
Gravel	None	10000	35%

Soil properties for the Brookside (Br), Guernsey (Gs), Melvin (Mh), Moshannon (Mo), Richland (Rc), Steinsburg (St), Vincent (Vn), Wellston (Wd), Westmoreland (Wl), and Fairpoint (Fa and Fb) soil series were taken from Athens County. More site-specific parameter estimates for hydrologic properties in Table 2.19 were discerned from the Meigs County soil survey (Gillmore *et al*, 1991) where possible.

2.8.9.3 Reference Definitions for Soil Properties

The following descriptions and definitions are taken directly from the soil surveys (Lucht *et al*, 1985) and are provided to allow for comparison of permeability, water capacity, hydrologic soils group, and soil pH, among all soils represented within Leading Creek.

- **Soil Permeability –** The ability of the soil to transmit water downward through its profile, in inches per hour through saturated soil.

- **Available Water Capacity (or Available Moisture) –** The capacity of soils to hold water available for use by most plants. It is expressed as the difference between the amount of soil water at field moisture capacity and the amount at wilting point.

- **Hydrologic Soils Groups –** The degree to which bare soil permits infiltration. Refers to soils grouped by the their run-off characteristics, independent of slope and vegetative cover.

- **Soil Reaction (or Soil pH) –** Soil reaction is a measure of the acidity or alkalinity of the soil, expressed in pH values.

Table 2.14 - Soil Permeability (Lucht *et al*, 1985)

Soil Permeability Rating`	Inches/hour
Very Slow	<0.06
Slow	<0.2
Moderately Slow	<0.6
Moderate	<2
Moderately Rapid	<6
Rapid	<20
Very Rapid	>20

Table 2.15 - Available Soil Water Capacity (Lucht *et al*, 1985)

Moisture Capacity Rating	Inches/60 Inch Profile	Inches/Inch Profile
Very Low	0-3	0-0.05
Low	3-6	0.05-0.1
Moderate	6-9	0.1-0.15
High	9-12	0.15-0.20
Very High	>12	>0.2

The data summarized for describing particle size diameter and other properties for each respective soil series was based on (1) surface layer (or topsoil) characteristics, and (2) the gentlest (common) slope available for the county survey. For the reclaimed Pinegrove (Pu), the Upshur silty clay loam portion of the Upshur-Gilpin (Ug) complex was used for physical characteristics. The actual Pinegrove (Pu) surface layer was formed from a mixture of nearby fine earth material and fragments of sandstone, siltstone, and shale derived from surface mining operations (Gillmore *et al*, 1991). For the Pinegrove (Pn), the Gallia Pinegrove (Pg) Sandy

Loam was used for physical characteristics. The Pinegrove (Pn) may have higher sand content and less silt/clay than shown.

Table 2.16 - Hydrologic Soil Group Rating (Lucht *et al*, 1985)

Hydrologic Soil Group	Run-Off Potential	Description
A	Low	High infiltration rate when wet; low run-off potential; typically deep, well drained sandy or gravelly soils
B	Modest	Moderate infiltration rate; moderately well drained; modest run-off potential
C	Moderate	Slow infiltration rate; somewhat poorly drained; moderate run-off potential.
D	High	Very slow infiltration rate; high run-off potential; clay layer at or near surface, permanent high water table, or shallow soil laid over impervious bedrock

Table 2.17 - Soil pH Rating Scale (Lucht *et al*, 1985)

Soil Reaction Rating Scale	Soil pH
Acidity Rating	
Extreme	<4.5
Very strong	<5
Strong	<5.5
Medium	<6
Slight	<6.5
Neutral	**6.6- 7.3**
Alkalinity Rating	
Mild	>7.4
Moderate	>7.9
Strong	>8.5
Very Strong	>9.1

If a complex is more prevalent (e.g. Ug) the respective complex is used for individual series (e.g. Gh and Ub) shown below. To avoid confusion, in soil series tables, the textures stated represent the individual series discussed. Modest differences may be noted for the actual individual series. The watershed dominant upland soil, the Upshur-Gilpin (Ug) is a silty clay loam. The Upshur and Gilpin series shown below represent data from the Ug complex. In Meigs County the primary series Upshur (Ub) is listed as a silt loam due to a 1” topsoil cover on these unmixed areas. The complex Ug occupies strongly sloping to steep terrain where the actual complex topsoil for individual series is a mix of original topsoil and subsurface materials.

Topsoil was assumed except for the USLE erosion factor T that applies to the entire soil profile. For the sieves given, data represents the percentage passing. Sieve sizes are given by #4= 4.75mm, #10 = 2 mm, #40 = 0.42 mm, and #200 = 0.074mm (Yang, 1996). Depth to bedrock represents the lowest portions of the substratum. Bedrock depth represents a minimum average in most cases and may be greater than shown (Lucht *et al*, 1985).

Table 2.18 - Leading Creek Soil Series – Particle Size Diameter
(Feusner *et al*, 1989, Lucht *et al*, 1985)

Soil Series Name	% > 10"	% > 3-10"	% < #4	%< #10	%< #40	%< #200	% Clay
Undisturbed							
Aaron (Aa)	0	0	94-100	95-100	85-100	70-90	10-27
Brookside (Br)	No data	0-5	90-100	80-100	70-100	55-90	18-27
Chagrin (Cg)	0	0	95-100	85-100	80-100	70-90	10-27
Elkinsville (Ek)	0	0	100	100	90-100	70-90	7-18
Gallipolis (Gb)	0	0	95-100	95-100	90-100	70-100	15-27
Gilpin (Gh)	0	0-5	80-95	75-90	70-85	65-80	15-27
Guernsey (Gs)	No data	0-2	90-100	80-100	75-95	70-90	13-27
Keene (Ke)	No data	No data	No data	No data	No data	No data	No data
Licking (Lk)	0	0	95-100	95-100	90-100	70-90	15-27
Melvin (Mh)	No data	0	95-100	90-100	80-100	80-95	12-17
Moshannon (Mo)	No data	0	95-100	95-100	90-100	70-95	15-32
Newark (Nk)	0	0	95-100	90-100	80-100	55-95	7-27
Nolin (No)	0	0	100	95-100	90-100	80-100	12-35
Omulga (Om)	0	0	95-100	90-100	85-100	65-90	12-18
Orrville (Or)	0	0	100	90-100	85-100	60-80	12-27
Rarden (Ra)	0	0	100	95-100	90-100	85-95	17-27
Richland (Rc)	No data	0-10	90-100	80-95	70-95	50-90	15-27
Steinsburg (St)	No data	0-5	95-100	90-100	65-90	35-70	10-20
Taggart (Ta)	0	0	100	100	90-100	70-90	12-20
Upshur (Ub)	0	0	95-100	95-100	90-100	80-95	27-35
Vandalia (Va)	0	0-5	80-100	75-100	70-95	50-90	20-35
Vincent (Vn)	No data	0	100	100	95-100	80-95	20-40
Wellston (Wd)	No data	0	95-100	90-100	85-100	70-95	13-27
Westmoreland (Wl)	No data	0	85-100	80-100	75-95	60-95	15-30
Disturbed							
Dumps, Mine (Dp)	No data	No data	No data	No data	No data	No data	No data
Fairpoint (Fa)	No data	0-15	90-100	80-100	70-100	50-90	18-27
Fairpoint (Fb)	No data	5-20	55-90	45-85	40-85	35-80	27-35
Kyger (Ky)	0	0	85-100	80-100	40-70	10-30	4-12
Pinegrove (Pn)	0	0-5	85-100	80-100	40-85	15-40	2-10
Pinegrove (Pu)	0	0	95-100	95-100	90-100	80-95	27-35

In reference to geology and soils, it is important to develop a context for the environmental setting under assessment (USEPA, 1996c, 1998). The data presented lays out preliminary evidence of what might be expected and what may be important in relating the physical and chemical characteristics of the landscape to ecological response. Digital data describing soils encountered in Leading Creek was not generally available, but would of course greatly improve upon the any study's ability to model hydrologically driven phenomena (Arnold *et al*, 1990, 1995; Bicknell *et al*, 1993; Donigian *et al*, 1995).

Table 2.19 - Leading Creek Soil Series– Hydrologic Properties
(Feusner *et al*, 1989, Lucht *et al*, 1985)

Soil Series Name	Topsoil Depth (in)	Bedrock Depth (in)	Water Table Depth (ft)	Water Table Type	Perme-ability (in/hr)	Water Capacity (in/in)
Undisturbed						
Aaron (Aa)	0-8	65	1.5-3	Perched	0.06-0.20	0.19-0.23
Brookside (Br)	0-5	60	2.4-4.0	Perched	0.6-2.0	0.19-0.24
Chagrin (Cg)	0-12	>60 (77)	4.0-6.0	Apparent	0.6-2.0	0.2-0.24
Elkinsville (Ek)	0-9	>60 (80)	>6	--	0.6-2.0	0.22-0.24
Gallipolis (Gb)	0-11	>60 (74)	2.0-3.6	Apparent	0.2-2.0	0.20-0.24
Gilpin (Gh)	0-6	31	>6	--	0.6-2.0	0.12-0.18
Guernsey (Gs)	0-8	50	2.0-3.5	Perched	0.6-2.0	0.19-0.24
Keene (Ke)	0-8	48	2.0-3.0	Perched	0.2-2.0	0.10-0.15
Licking (Lk)	0-7	> 60 (72)	1.5-3.0	Perched	0.06-0.20	0.21-0.24
Melvin (Mh)	0-4	60	1	Apparent	0.6-2.0	0.18-0.23
Moshannon (Mo)	0-7	>60 (70)	4.0-6.0	Apparent	0.6-2.0	0.20-0.24
Newark (Nk)	0-9	>60 (62)	0.5-1.5	Apparent	0.6-2.0	0.15-0.23
Nolin (No)	0-9	>60 (82)	3.0-6.0	Apparent	0.6-2.0	0.18-0.23
Omulga (Om)	0-9	> 60 (82)	2.0-3.5	Perched	0.6-2.0	0.22-0.24
Orrville (Or)	0-4	>60 (80)	1.0-2.5	Apparent	0.6-2.0	0.18-0.22
Rarden (Ra)	0-6	32	1.5-3.0	Perched	0.06-0.20	0.21-0.24
Richland (Rc)	0-10	>60 (80)	3.0-6.0	Apparent	0.6-2.0	0.16-0.20
Steinsburg (St)	0-3	36	>6	--	2.0-6.0	0.10-0.14
Taggart (Ta)	0-8	>60 (80)	1.0-3.0	Apparent	0.06-2.0	0.22-0.24
Upshur (Ub)	0-4	53	>6	--	0.06-0.20	0.12-0.16
Vandalia (Va)	0-6	> 60 (80)	4.0-6.0	Perched	0.20-0.60	0.12-0.18
Vincent (Vn)	0-8	> 60 (80)	2.0-4.0	Perched	0.06-0.20	0.20-0.24
Wellston (Wd)	0-8	48	>6	--	0.6-2.0	0.18-0.22
Westmoreland (Wl)	0-9	45	>6	--	0.6-2.0	0.16-0.20
Disturbed						
Dumps, Mine(Dp)	No data	No data	No data	No data	No data	No data
Fairpoint (Fa)	0-10	60	>6	--	0.6-2.0	0.14-.020
Fairpoint (Fb)	0-5	60	>6	--	0.2-0.6	0.06-0.15
Kyger (Ky)	0-19	>60 (80)	-2 to 1	Ponding	6-20	0.08-0.10
Pinegrove (Pn)	0-2	>60 (80)	>6	--	2-20	0.09-0.13
Pinegrove (Pu)	0-5	>60 (80)	>6	--	0.20-0.60	0.12-0.16

The Pinegrove(Pu) permeability is rapid in the substratum, a shallow topsoil layer is described.

Table 2.20 - Leading Creek Soil Series–Miscellaneous Properties
(Feusner *et al*, 1989, Lucht *et al*, 1985)

Soil Series Name	Soil Reaction (pH)	% Organic Matter	USLE K	USLE K_f	USLE T	Hydro-logic Group
Undisturbed						
Aaron (Aa)	4.5-7.8	1-3	0.37	0.37	3	C
Brookside (Br)	5.6-7.8	1-4	0.37	No data	5	C
Chagrin (Cg)	5.6-7.3	2-4	0.32	0.32	5	B
Elkinsville (Ek)	5.6-7.3	0.5-2	0.37	0.37	5	B
Gallipolis (Gb)	5.1-7.3	1-3	0.37	0.37	5	C
Gilpin (Gh)	3.6-5.5	0.5-4	0.32	0.43	3	C
Guernsey (Gs)	4.5-6.5	1-3	0.43	No data	3	C
Keene (Ke)	No data	No data	No data	No data	No data	No data
Licking (Lk)	4.5-6.0	1-3	0.43	0.43	3	C
Melvin (Mh)	5.6-7.8	0.5-3	0.43	No data	5	D
Moshannon (Mo)	5.6-7.3	1-3	0.37	No data	5	B
Newark (Nk)	5.6-7.8	1-4	0.43	0.43	5	C
Nolin (No)	5.6-8.4	2-4	0.43	0.43	5	B
Omulga (Om)	4.5-7.3	0.5-2	0.43	0.43	4	C
Orrville (Or)	5.1-7.3	2-4	0.37	0.37	5	C
Rarden (Ra)	3.6-6.5	1-3	0.43	0.43	2	C
Richland (Rc)	5.1-7.3	1-3	0.37	No data	5	B
Steinsburg (St)	4.5-5.5	0.5-3	0.28	No data	2	C
Taggart (Ta)	4.5-7.3	1-3	0.37	0.37	5	C
Upshur (Ub)	4.5-6.5	0.5-3	0.37	0.37	3	D
Vandalia (Va)	4.5-6.0	1-3	0.37	0.43	4	D
Vincent (Vn)	5.1-7.3	1-3	0.43	No data	3	C
Wellston (Wd)	5.1-7.3	1-3	0.37	No data	4	B
Westmoreland (Wl)	4.5-6.0	1-4	0.37	No data	3	B
Disturbed						
Dumps, Mine (Dp)	No data	No data	No data	No data	No data	No data
Fairpoint (Fa)	5.6-7.3	0.5-2	0.42	No data	3	C
Fairpoint (Fb)	5.6-7.3	<0.5	0.37	No data	5	C
Kyger (Ky)	3.6-5.5	0-0.5	0.17	0.20	5	B
Pinegrove (Pn)	3.6-5.5	0.5-1	0.24	0.28	5	A
Pinegrove (Pu)	4.5-6.5	0.5-3	0.37	0.37	5	D

2.9 HYDROGEOLOGY, HYDROLOGY, AND HYDRAULICS

The Leading Creek Watershed lies entirely within the unglaciated Allegheny Plateau region of Ohio, where the areas have been extensively dissected by drainageways (Feusner *et al*, 1989; Lucht *et al*, 1985; Gillmore, 1991). As it is important to get a feel for the landscape's basic geology and hydrology, it can also be beneficial to further examine the transport and depositional elements of the idealized fluvial model for Leading Creek (Diplas, 1997; Schumm, 1977). It is helpful to place hydrologic analysis also in the context of larger hydrologic units that contain a given watershed under investigation.

2.9.1 Historic Drainage Patterns of the Region

In the north portions of the watershed in Albany, a more unique feature in Athens County still prevails, with extensive preglacial terraces sitting atop gently sloping high level land. Athens County otherwise is usually described by its prominent landscape of hills, narrow ridgetops, and stream valleys drained by the Hocking, Monday, Sunday, and Federal Creeks. Federal Creek is shown in Figure 2.3 and was monitored here as reference site for comparison to Leading Creek.

Leading Creek's northern and western headwater drainages, above and including Dexter Run's headwaters, are also direct remnants of the Teays River drainage system. Preglacial terraces characterize these areas, with valley floors in Athens and Meigs County ranging 820 feet to 700 feet AMSL. The remaining valley floors of these tributaries along with Leading Creek's other tributary headwater valleys in the southern and western subwatersheds are found at lower relief, initiating at ranges of 680 feet to 760 feet AMSL (Lucht *et al*, 1985).

The Teays River was an extensive network of streams draining northwest and flowing from the Carolina region towards Wisconsin (Lucht *et al*, 1985). With the advance of glacial ice the Teays River system was dammed by glacial deposits, forming an extensive lake system. Glacial meltwater and outwash (Feusner *et al*, 1989; Lucht *et al*, 1985) produced Leading Creek and forms the existing gently sloping valley network seen today. Less than 100km to the northwest of Leading Creek in Fairfield County, Ohio, the northwestern headwater areas of the Hocking River watershed are reported to include small amounts of glacial till resulting from the southernmost extension of the last glacier (Lucht *et al*, 1985).

Unglaciated itself, Leading Creek is a remnant of the original Teays River valley. Portions of the lower subwatershed areas of the Leading Creek Watershed are typified by long and steep hills, steepest adjacent to the Ohio River (Feusner *et al*, 1989). Similar long and steep hill features are observed in the surrounding outside areas in eastern Meigs, and eastern Gallia Counties, and along the Ohio River.

2.9.2 The Ohio River Basin and Leading Creek

The regional hydrologic units encompassing the Leading Creek Watershed and the surrounding area have been extensively dissected by drainageways leading to the Ohio River. To place Leading Creek within the context of its parent hydrologic basin unit, the Ohio River Basin, the following facts and data were summarized directly from the Ohio River Valley Water Sanitation Commission (ORANSCO) website reports (CVC, 1998):

- ✓ *The Ohio River begins in Pittsburgh, Pennsylvania, at the confluence of the Allegheny and Monongahela Rivers near Three Rivers Stadium.*

- ✓ *The Allegheny River flows 325 miles and drains 11,778 square miles, flowing North from Coudersport, PA, through Olean, NY, before turning south and flowing to the Ohio River at Pittsburgh.*

- ✓ *The Monongahela River, which flows 128 miles and drains 7,386 square miles in northern West Virginia, southwestern Pennsylvania and northwestern Maryland. The "Mon," flows north to meet the Allegheny River in Pittsburgh to form the Ohio River.*

- ✓ *The Ohio River Basin drains 203,940 square miles and flows 1,579 kilometers or 981 miles to Cairo, IL, where it empties into the Mississippi River. The first 40 of those miles are in Pennsylvania, with the remaining reaches touching five other states including Ohio, West Virginia, Kentucky, Indiana, and Illinois.*

The Leading Creek catchment falls within the Upper-Ohio Shade Subbasin. One may note on terminology that the latter cataloging unit itself is labeled by USEPA as "Watershed ID" = 05030202 (Seaber *et al*, 1987). The Upper-Ohio Shade Unit is 1403 square miles with a perimeter of 227 miles. The Upper-Ohio Shade Unit drains the Little Hocking River, Shade River, Leading Creek, Kyger Creek, and on the West Virginia side, the Little Kanawha River, Sandy Creek, Lee Creek, Pond Creek, Mill Creek, and OldTown Creek (See Figure 2.1). There are 32 stream segments defined in the Upper-Ohio Shade unit according to USEPA's River Reach File. According to USEPA, 16% of the unit's streams have been surveyed (USEPA, Surf Your Watershed website). Leading Creek Watershed makes up 10.8% of the Upper-Ohio Shade hydrologic unit.

2.9.3 The Ohio River, Leading Creek, and a Glass of Water

With respect to Leading Creek's individual contribution, the percentage by area of the Ohio River at its confluence point with Leading Creek in Middleport, Ohio is 0.54% (28,000 square miles total). At the discharge point of the Ohio River into the Mississippi River, Leading Creek represents 0.074% of the Ohio River. The Ohio River receives 89.4% of its flow from tributaries draining > 1000 square miles or more in area (i.e. tributaries that confluence directly with its mainstem) (CVC, 1998). Of the remaining flow, 6% is from tributaries draining between 75 to 1000 square miles, and 4.5% comes from areas in close proximity to the river channel. Leading Creek, in the minority 10.6% of smaller tributaries that confluence the Ohio River, drains ~150.2 miles^2 or ~38km^2.

According to ORANSCO, the Ohio River is a drinking water source for 3 million people (CVC, 1998). The average residential consumer of Ohio River water downstream of Leading Creek might utilize 60gal/day personal use over a lifetime of 70 years. Of this amount 12% may come directly from faucets/taps, and 1/3 might be used for dishes, baths/ and showers (Metcalf & Eddy, Inc., 1991). On average that person will eventually consume a total of between 1120 to 8300 gallons of water drained from Leading Creek, depending on where they live from Cairo, Illinois to Gallipolis, Ohio. Derived from Leading Creek, on average the same consumer will use 0.04 to 0.32 gallons total per day, or roughly 6 to 41 ounces/day, of which 0.7 to 5 ounces might be used as tap water. Thus, a consumer who drinks two 8 oz. glasses of Ohio River water a day will ingest 0.01 to 0.09 ounces/day of water from Leading Creek or ½ teaspoon per day.

2.9.4 Preliminary Hydrologic-Hydraulic Assessment

The perennial and non-perennial drainages of the Leading Creek Watershed represent a riverine environment that drains moderate to steeply sloping uplands and gently sloping lowlands. The landscape is typified by steep and very steep rugged hills with narrow and broad ridgetops.(Gillmore *et al*, 1991). Mainstem slopes of the stream bed range from 0.01-0.0001 ft./ft (50 to 0.5 ft/mile or 1% to 0.01%), from headwaters below Albany, Ohio to the mouth of the Ohio River, respectively. Qualified by attempts to describe any "reach", "cross-section", or "site", definition as such can oversimplify the potentially diverse heterogeneity that exists within even one cycle of a pool-riffle-run habitat (Rankin, 1989; OEPA, 1989; USEPA, 1989, 1996c; Yang, 1996).

2.9.5 Stream Energy Slope and Bed Profile

Longitudinal profiles of energy slope on Leading Creek's mainstem follow an expected characteristically concave-up shape (Diplas; 1997; Dunne *et al*, 1978). As seen from the USGS 7.5-minute series quadrangles and Figure 2.4, a significant transition in slope occurs in a 4km stretch of the mainstem, beginning around Jones Road (Township Road 3) at river kilometer (RKM) 48 down towards the town of Carpenter, Ohio at RKM 44. Above this transition area the stream exhibits bedrock and cobble sediment features, where a "gravel-bed" description is hydraulically accurate.

Below this "headwater" or steeply graded portion of the mainstem, smaller gravel-size particles and eventually sands and finer materials begin to significantly dominate the stream course bed. This is due to natural sorting processes (Andrews *et al*, 1993; Diplas, 1997; Lisle, 1989; Milhous; 1982; Schumm, 1977; Vanoni, 1977). Under the influence of man, particle size can also decrease (become finer) due to increased fines loading from anthropogenic disturbances of the ecosystem such as AML and AG sedimentation. Such sediment loading, derived in most part from the landscape, can be provided by activities such as farming, mining, and dredging (Adams *et al*, 1980; Curtis, 1973, 1979; Diplas, 1992b, 1994, 1997; Lisle, 1989; Miller, 1988; Waters, 1995; USDA, 1983, 1985, 1998). In some cases these activities can overwhelm otherwise naturally stable geomorphology, decreasing natural carrying capacity of the fluvial channel (Leopold *et al*, 1964; Rosgen, 1996).

On the mainstem, below the toe area of the headwater region slope around Carpenter, Ohio and down into Rutland, Ohio, Leading Creek maintains a weaker pool-riffle configuration with alternating bars, tributary bars, and some middle bars (OEPA, 1989; Yang, 1996). The mainstem eventually transitions to more plane bed reaches between tributaries, initially dominated by sands below Lasher Run around RKM 16, and even more so below Little Leading Creek at around RKM 13. The lower AML primary tributaries up to and including Lasher Run are sometimes referred to here as the "lower mining belt". Stream courses within the lower mining belt area are characterized by sand beds with small-amplitude bed form ripples, alternating bars, and increased fine sediment storage (Leopold *et al*, 1964; Vanoni, 1977; Yang, 1996).

Aside from natural sorting processes, mainstem inundation by sands below RKM 16 is attributed in most part to the more weather resistant sandstone cap-rock spoils from abandoned strip mine lands (ASML). Other notes of hydraulic interest, there are several areas with beaver activity and associated elongated pool structures between RKM 48 and RKM 32, shown in Figure 2.4.

Elongated pool structures in this region appear in some cases to coincide with major transitions in soil association and floodplain-channel soil textures, for instance at RKM 32 in Dexter, Ohio.

2.9.6 Navigation/Hydroelectric Operations Along the Ohio River

Leading Creek enters the Ohio River at River Mile 254.2, below Pittsburgh, PA. The Ohio River influences the Leading Creek Watershed. The Ohio River is itself controlled by navigational dams. The lock/dams in some cases have side-by-side hydroelectric power generation facilities. Near Leading Creek, the Ohio River is controlled by a hydroelectric lock/dam at Racine (Ohio River RM 382.2) and at the Robert C. Byrd Lock, near Gallipolis, Ohio (Ohio River RM 449.3). In total the Ohio River has 20 dams with a minimal navigational depth of 9 feet available for commercial navigation. The Ohio River also has 49 power-generating facilities equivalent to 6% of the entire U.S. power output capacity (CVC, 1998).

Locks work on the principle of temporarily containerizing water, and either lifting ships placed within that container (by adding more water) or lowering them (by releasing water). Upstream water is of course efficiently used by gravity filling of the container. Water is released downstream, typically with overall minimal fluctuations to mean pool levels. Most hydropower plants release on their own schedule, based on optimal power generation, only having to maintain a prescribed minimum navigational depth in the upstream pool at all times (e.g. 9 ft).

The Racine Ohio Lock with normal (upstream) pool elevation at 170.7m above mean sea level (AMSL- National Geodetic Vertical Data) was placed into operation in 1967. The Robert C. Byrd Lock downstream was placed into operation in 1937 with pool elevation at 164m AMSL (CVC, 1998). This is the pool that interacts directly with the mouth of Leading Creek. The Byrd Lock is reportedly one of the busier locks on the Ohio River, and was modernized this decade.

The Racine power facility is actually operated via remote control by a management firm in Roanoke, Virginia. Each power facility is typically operated independent from one another, where the Army Corp of Engineers maintains regulatory authority over the river and its dams, and runs the actual day to day lock operations. The locks and hydropower facilities typically monitor daily rainfall, dissolved oxygen, and stage/flow (personal communication with Paul Smith, Racine Ohio Power Plant Engineer).

2.9.6.1 Influence of the Ohio River on Leading Creek

The hydraulic influence of the Ohio River is evident under all flow conditions at Middleport, Ohio at the mouth of Leading Creek. The Ohio River interaction with Leading Creek increases the complexity of hydraulic relations. Backwater influences of the Ohio River of course reach further upstream on Leading Creek with increased stage along the Ohio River. As a practical matter, the scope of this study was limited to directly evaluating flow/transport characteristics mostly for non-backwater events only. Hydrologic assessment focused on otherwise natural drainage features above the mouth. Backwater was evaluated by defining the relative patterns and associated areas of influence. For assessment of lower mainstem and tributaries stations under the potential influence of the Ohio River, backwater must be accounted for to ascertain whether conditions influence hydraulic data sets.

Due to the relative sizes of Leading Creek and the Ohio River, there would be significant backwater effects on Leading Creek during major storms even if there weren't any locks or dams. Aggradation and degradation are likely side effects of control on the Ohio River by locks and dams (Lane, 1955). Aggradation above and degradation below each respective control point on the Ohio River itself would be expected. This could balance out at the confluence of Leading Creek, itself about ½ way between lock control points on the Ohio River.

2.9.7 Low to Moderate Flows In Leading Creek

At low flow many smaller tributaries can run dry in Leading Creek. The two Southern Ohio Coal Company (SOCCO) coal mine plant discharges (Meigs Mine No. 2 and No. 31) represent significant groundwater diversions during dryer periods (contributing up to 50 to75% of flow at mouth, and 300% at Parker Run confluence). These features would be important for consideration in their ability to transform the natural fluvial system that in turn likely will have a large influence upon biotic integrity (Karr, 1991; Leopold *et al*, 1964; Schumm, 1977; USEPA, 1996c). According to Ohio EPA, the two plants produce 5.5 to 6 million tons of coal annually

A natural feature of alluvial streams is that they tend to meander through narrower potions of the channel bed at extreme lower flows. Bed form resistance (ripples, dunes, bars, etc.) also optimizes water retention during low flow in the form of pools and ultimately as groundwater traveling slowly though bed sediments, all at an extreme benefit to biota (Allan, 1995; Diplas; 1997; Fetter; 1994; Vanoni, 1977; Yang; 1996).

Leading Creek under most average to moderate flow conditions maintains a naturally occurring weak riffle-pool configuration. At low flow this still holds partly true, but the configuration is further weakened towards more homogenized flat bed qualities by the addition of fine sand materials derived from AML. Effective water depth can be severely reduced by added loads of sandy materials in many areas along the mainstem and tributary-stems. This can be exacerbated by smaller storm pulse events that deliver finer sediments to the channel, but do not completely carry them away, or that only move them partially downstream. Non-flooding periods of flow represent opportunity for infiltration of gravel by fines < 0.125mm, building of bars, and the filling of pools where inundating sedimentation can also be severe for larger sand fractions with excess sediment supply (Diplas, 1992b, 1994, 1997; Lisle, 1989; Waters, 1995).

2.9.7.1 Riffle Bar Health

It is well established that typically aquatic organisms (insects and fish) show a reduced capacity of production in response to increases in fine bed material (Diplas, 1992b, 1994; Richards *et al*, 1994a, 1994; Rosenberg *et al*, 1986, 1993; Shirazi *et al*, 1981). Engineering flume studies show that under shear-stress conditions below effective threshold pavement mobility, increases in fines loading accumulate to varying degrees in gravel beds. Pool gravel will first begin to fill with fines, moving eventually towards the bar and its various distinguishable parts (e.g. bar head, bar tail, etc) (Diplas, 1992b, 1994). It has been established that mobile pavement conditions are needed to clean fines from both the pavement and subpavement zones of gravel bars (Diplas, 1994). Other discussions in literature also exist for evaluating appropriate flushing flows, frequency, and duration (Milhous, 1982, 1998; Wesche *et al*, 1980; Wick *et al*, 1998).

2.9.8 Flooding in Leading Creek

In just the last few years an unusually high frequency of strong storms and associated flooding have hit the Leading Creek Watershed, and surrounding areas, compared to trends earlier this century. Heavy storms with excessive flooding were recorded in the area on Mothers Day, May 12 1995, January 16, 1996, May 12, 1996, March 1, 1997, and January 13, 1998, among other notable bankfull events. March 1997 was the most significant storm. During monitoring between 1996 and 1997, Leading Creek experienced 9 bankfull flows, where normally one such storm event every 1.6 years might be expected (Rosgen, 1996).

In gravel bed and sand dominated streams, such as Leading Creek, it has been estimated that 99% of sediment is carried during storm events which comprise less than 1%-3% of the time (Diplas, 1997; Leopold *et al*, 1964). A perspective of flooding is useful if not requisite in understanding the boundaries of sediment transport. The following discussion offers some insight as to how flooding affects residents and businesses directly, and gives similar insight as to how they affect flooding.

As one example of why more flooding occurs now, an observation was recorded where County road crews were cleaning up post-flood soil debris after the March, 1997 storm. The materials had slipped from a bank cut onto the roadway. An excavator was being utilized to clear about 10 cubic yards of materials from the roadway, dumping them over the guardrail towards the streambed area below. This can be classified as poor floodplain management. At the same time, the example should not be used to take emphasis away from the more widespread and severe impacts associated with agriculture and AML.

Flooding is often the most frequently noted source of resident-stress encountered in streamside discussions held with inhabitants of Leading Creek. Local perception and discussion about flooding is obviously elevated due to the string of recent floods. Flash flooding can be severe throughout the basin depending upon rainfall intensity and duration. Based on discussions with several older residents, most today hold the belief that flash flooding has increased in recent decades for the same storm conditions, relating their observation of decreased channel capacities due to sedimentation in many locations of the creek.

Under moderate flooding, many bridge structures are inundated. Current transportation network design for the most part does appear to allow for emergency access to most mainstem areas under moderate flooding. Under severe flooding, vehicle access to the central and southern areas can be severely restricted for up to 24-48 hours, and longer depending on backwater flooding along the Ohio River. There are two primary exceptions with increased flooding frequency. The first is below Dexter on CO-10 to its intersection with Ward Road (~RM 26 to RM 27). The second is below Rutland on CO-03 (e.g. particularly below Twin Bridges Crossing), the latter is an aspect of Ohio River backwater, where natural discharges on the mainstem below Rutland, Ohio are not highly significant to most residents or farmers

2.9.8.1 Bankfull and Other Flood Recurrence Intervals

The Dexter area roads are flooded at just above bankfull condition with more limited duration for bankfull to moderate flooding (8 – 24 hours). For this area of the country, a typical long-term bankfull design recurrence interval of 1.5 to 2 years would be expected to prevail (Diplas, 1992b; Leopold *et al*, 1964; Rosgen, 1996). Bankfull discharge estimates are sometimes seen also as an effective discharge condition or channel forming condition, or dominant discharge condition (Goodwin, 1998; Hey *et al*, 1998). All in some manner represent the systems composite morphogenetic trait. Thus, the Town of Dexter, Ohio may expect on average to exceed a significant floodstage, in this case bankfull condition, about once every 1 to 2 years. Annually, the area below Rutland (Leading Creek RKM 14 and below) is subject to frequent episodes of inundation by the Ohio River. Flooding on Leading Creek will occur at pool levels below "floodstage" on the Ohio River itself (i.e. with respect to the Town of Pomeroy, Ohio floodstage).

According to regional estimates made across the U.S., the anticipated mean annual flood on Leading Creek would be ~ 4000 - 6000 cfs. For 10-year flood estimates (e.g. 10-year storm) approximately 9000 cfs might be expected (Leopold *et al*, 1964). These estimates appeared to be on the high side of actual rates measured based on initial site reconnaissance data, but were reasonable order of magnitude predictions.

In summary, it is important to put into context that degree of flooding has likely been increased in Leading Creek for this time period and climate due to channelization, floodplain encroachment, and overall increased sediment and water loads from the land surface. These impacts in turn can all increase stream bank erosion for this geology type and morphology (Lane, 1955; Leopold *et al*, 1964; Rosgen, 1996). Sedimentation of Leading Creek affects flash flooding in all areas, but likely has minimal to insignificant impacts upon backwater flooding extent; longitudinally, laterally, and vertically.

2.9.8.2 The Flood of March 1997

The flood of March 1997 at mid-basin measured ~5" of rain in less than 24 hrs, close to a 100-year flood event. According to SOCCO personnel, the Leading Creek's 100-year flood recurrence interval is somewhere on the order of ~6" of rain in 24 hrs. This 6" figure is an estimate only, but helpful in describing the watershed. As an educated guess, the existing rail line that travels the valley floor from the Ohio River up through to the headwaters at Albany, Ohio has tracks set at approximately the 100-year flood plain. The tracks were covered in some places during the Flood of 1997, but for the most part remained dry in other areas. In these extreme situations all watershed rail activity shuts down for several days.

Analysis of various flow regimes can be beneficial to understanding relationships between the landscape and biotic integrity. Low flow and flood flows are typical design conditions of concern to both scientists and engineers (Bedient *et al*, 1992; Karr, 1991; USEPA, 1996c; Yoder *et al*, 1996)

2.10 LAND USE/COVER

The following data was reviewed to first assess general trends in land cover and land use across the Leading Creek Watershed. Shown in Table 2.21, Meigs County hosts 96% of the Leading Creek watershed, where the Leading Creek comprises 33% of Meigs County, in primarily the western portions. On average, it was estimated that Federal lands comprise < 0.5% of Leading Creek's total land surface.

Table 2.21 - Federal, Non-Federal, and County Lands (USDA, 1983)

Ohio County	Federal Land (ha)	Non-Federal Land (ha)	Total Land (ha)	Land Draining to Leading Creek (ha)	Relative % of Leading Creek Drainage Area
Athens	4,310	126,230	130,540	1,052	2.7%
Meigs	~ 0	112,886	112,800	37,217	96%
Gallia	4,100	117,898	122,000	537	1.4%

Above County totals were derived from soil survey data. Federal land was estimated from non-Federal land totals. Other estimates were derived from the Leading Creek GIS developed through this work.

2.10.1 Human Settlement and Demographics

According to (Feusner *et al*, 1989; Lucht *et al*, 1985; Gillmore *et al*, 1991):

- Athens County originally included 5 townships now residing within Meigs County.
- The first settlers in the region were primarily from New England, and were veterans of the Revolutionary War.
- Communities were often organized into townships and villages, the organizational structure that persists today. In early settlement, townships often set land aside for a school and for religious institutions.

2.10.1.1 Political Boundaries

- Small portions of Lee and Alexander Townships are included at the north end of the Leading Creek Watershed in Athens County. Otherwise Columbia, Scipio, Salem, Rutland, Chester, and Salisbury Townships span most other areas of Leading Creek.
- A small portion of Cheshire Township in Gallia County intersects the southern tip of the watershed.
- Representing incorporated localities, parts of Albany Village, Middleport Village, and all of Rutland Village lay within the Leading Creek Watershed.
- Unincorporated towns along the mainstem trunk from northwest to southeast include Carpenter, Dyesville, Dexter, Langsville, and Hobson at the creek mouth. Off the upper mainstem trunk is Valley Ford, Welsh, and Hanesville. Along Little Leading Creek,

unincorporated towns include Harrisonville, New Lima (as well as incorporated Rutland Village). Along Thomas Fork, unincorporated towns include Laurel Cliff, Thomas, and Bradbury.

- According to USEPA's Surf Your Watershed Website database (based on US Census data), no known Indian Tribes are known to occur within the Upper-Ohio Shade Cataloging Unit.

2.10.1.2 Human Population

- Middleport is the largest single community in Meigs County with 2,725 residents.

- Based on Meigs County Chamber of Commerce data, the 1980 population census count in Athens County was 53,311 and in Gallia County it was 30,098. 1990 census for Meigs County indicated 22,987 residents.

- Using Meigs County data, on average (based on area) approximately 7500 to 7800 residents may live within the watershed perimeter. A census of approximately 12,400 was estimated based on area and population statistics derived from Upper-Ohio Shade HUC water supply data made available from USGS.

- Counting population changes between 1990 and 1994 occurring within townships and villages encompassing Leading Creek, increases in population of 1049 residents were observed (on average) or equivalently a 4.7% increase was noted from the start of this decade.

- Farming occupation is on the decline in Leading Creek while residential uses are on the incline.

2.10.2 Regional and Subregional Assessment of Land Use/Cover Trends

As reported by the National Watershed Network, land use/cover trends for the Upper – Ohio hydrologic unit (HUC: 0503), were given by data found in Tables 2.22 and 2.23:

Table 2.22 - Land Use/Cover Trends Upper-Ohio Hydrologic Unit (0503)
(National Watershed Network website information – October, 1998)

Cropland	**Grazing**	**Pasture**	**Forestry**	**Mining**	**Urban**	**Total**
33%	17%	15%	31%	2%	4%	102%

The urban rating above was assumed as it was obviously in error; actually listed as 40%.

Grazing above was interpreted as active pastureland, where a total of 32% would otherwise be assigned here to pasture. The source of the above geo-statistics was not known, but likely depended on coarse-scale geodata. Water uses for the Upper-Ohio subregion were listed as drinking water, fisheries, hydroelectric power generation, navigation, and recreation. Stresses and pollutants simultaneously noted by the regional studies indicated potential concerns of:

1) Bacteria,
2) BOD/DO,
3) Heavy Metal (s),
4) Nitrogen - Phosphorous,
5) Pesticides, and
6) Pathogens.

Table 2.23 - Land Use/Cover Indicators - Upper-Ohio Shade River Hydrologic (05030202)
(USEPA Surf Your Watershed website information - October, 1998)

Crop-Use	**Forestry**	**Urban**	**Total**
9%	96%	1%	106%

USEPA provided a note that overlapping of uses described may occur (e.g. sums > 100% are possible). The source of geo-statistics was not known.

The Ohio EPA (OEPA) has conducted biological and ecological assessments of various hydrologic units throughout Ohio. Those reports for Leading Creek and its reference watersheds are discussed in further detail here. The reports point-out that in the Western Allegheny Plateau (WAP) Ecoregion, less than 20% of the land is used for cropland and pasture which tend to be located along narrow steam valleys (OEPA, 1991a; USEPA, 1988). The most common sources of non-point pollution identified in referenced reports were associated with resource extraction, farming, and silviculture (OEPA, 1991a, 1991b). Sedimentation from urban and mining activities is known to be a significant impact from nonpoint sources (USDA, 1983, 1985, 1999; OEPA, 1991a, 199b). Second only to organic enrichment and dissolved oxygen depletion, sedimentation was recognized more recently as the second leading cause of impairment to streams and lakes from all sources identified by Ohio EPA in their 1994 Ohio Water Resource Inventory (Yoder *et al*, 1996).

2.10.3 County-Level Land Use/Cover Trends

Given Meigs County's dominant coverage of the watershed (96% by area), relative percentages of several land use categories in the county are introduced in Tables 2.24 and 2.25 with their respective erosion trends for comparison (Cherry *et al*, 1999; USDA, 1983).

Table 2.24 - 1979 Non-Federal Land-Use Data - Meigs County
(Cherry *et al*, 1999; USDA, 1983)

Cropland (ha)	**Pasture (ha)**	**Forest (ha)**	**Urban (ha)**	**Other (ha)**	**Total (ha)**
20,305	19,080	63,894	5094	4,512	112,800
18%	17%	57%	5%	4%	100%

The local Meigs County soil survey indicated coal mining, woodland management (silviculture), farming, and industry were the major land uses in 1990 (Gillmore *et al*, 1991). As seen from the analyses in Table 2.25, 20% of total land-use was deemed over erodible limits. Except for SOCCO coal extraction operations, other industry and commerce within the watershed boundaries appears to be minimal and light (Gillmore *et al*, 1991). Although the major land use in the watershed was forested land, cropland and pastureland accounted for 35% of total land use, where 35% of those land-uses were exceeding erodible limits for incident soils (USDA, 1983).

Table 2.25 - Land Eroded Over Tolerable Limits (T) - Meigs County
(Cherry *et al*, 1999; USDA, 1983)

Cropland		Pasture		Forest		Total	
Land Over T (ha)	***% Over T***	***Land Over T (ha)***	***% Over T***	***Land Over T (ha)***	***% Over T***	***Land Over T (ha)***	***% Over T***
7,178	35%	10,676	56%	5,250	8%	23,104	20%

Countywide land use/cover appears to fall within average regional-scale and subregional-scale assessments. Though mining trends were similar, agricultural trends indicated that Meigs County and Leading Creek have higher rates of farming and lower rates of woodland cover than its host Upper-Ohio Shade HUC does. At coarser scales, the opposite was noted for the Upper-Ohio HUC. The latter had a total of 65% of land uses attributed to agricultural.

For direct application to Leading Creek, the quality of regional and subregional land use/cover data listed above is limited by coarse-scale, which in general can be a problem in biological studies (Richards *et al*, 1994b). The land use/cover references presented do though provide useful confirmation that mining land-use trends and data appear as expected based on site walkovers, as do other trends in agriculture. Discussed further below, Leading Creek ranked high regionally with respect to associated potential impacts from surface mining, having 4.5% by area attributed to ASML, where regional trends average 2% (CVC, 1998). It is possible differences could simply be reflecting definitions of AML and inventory methods, which would be suspected to be underestimated for less quantitative, comprehensive inventories. Total near-surface abandoned underground mining was 4.1% by area in Leading Creek.

In comparison to countywide use rates, Leading Creek may actually maintain slightly higher levels of woodland and lower acreage set-aside for farmed land. According to local silviculturists commenting on land use/cover trends, reduction in farmland tracts have traded-off with in the increased conversion of farmlands to brushland and to 20-30 acre residential tracts and associated land-uses. This appears related in part to an increase in population and reduced economy in farming. Most workers in Meigs County appear to be employed outside of the watershed, except for those working for SOCCO. SOCCO has approximately 1050 employees.

2.10.4 Regional and Local Water Supply Trends

2.10.4.1 Upper-Ohio Shade Water Supply Data

The following water supply data was made available from USGS. The data is summarized for total water uses for the USGS HUC 05030202 - Upper Ohio-Shade - USGS Cataloging Unit. An equivalent ratio by area was also applied to estimate an “average” Leading Creek value (as unverified estimates only). From the data below, on average, Leading Creek inhabitants consume 1.4 million gallons of freshwater per day (MGD). In the subbasin 9% of public water supplies come from surface water, the remaining 91% are provided from groundwater supplies.

Data on water uses in the subregion is presented in Table 2.26. A total of 2266 MGD freshwater is used in the entire subbasin. The primary use of freshwater in the larger Upper-Ohio Shade hydrologic unit is for fossil-fuel thermoelectric/hydroelectric power generation at four facilities. Total withdrawals for these operations (e.g. presumably Ohio River surface water) are 2239 MGD. Approximately 34,000-gigawatt hours/year of production are realized. There are 111 wastewater treatment facilities in the hydrologic unit, 30 of which are public plants. The public treatment facilities return (treat) 16.2 MGD.

Table 2.26 - USGS 1990 Total Water Supply and Public Water Supply Data - Leading Creek Watershed and the Upper-Ohio Shade Hydrologic Unit (05030202)

Totals	**Units**	**Shade Unit**	**Leading Creek**
Ground-water withdrawals, fresh	MGD	25.42	2.74
Ground-water withdrawals, saline	MGD	0	0
Withdrawals, ground water	MGD	25.42	2.74
Surface-water withdrawals, fresh	MGD	2241.03	241.13
Surface-water withdrawals, saline	MGD	0	0
Surface-water withdrawals	MGD	2241.03	241.13
Fresh-water withdrawals	MGD	2266.45	243.87
Saline withdrawals	MGD	0	0
Withdrawals	MGD	2266.45	243.87
Reclaimed wastewater	MGD	0	0
Fresh consumptive use	MGD	174.53	18.78
Saline consumptive use	MGD	0	0
Consumptive use, total	MGD	174.53	18.78
Conveyance losses	MGD	0	0
Population			
Total population of the area	Thousands	115.46	12.42
Public Supply			
Population served by ground water	Thousands	91.87	9.89
Population served by surface water	Thousands	9.09	0.98
Total Population served	Thousands	100.96	10.86
Ground-water withdrawals, fresh	MGD	12.43	1.34
Surface-water withdrawals, fresh	MGD	0.6	0.06
Total withdrawals, fresh	MGD	13.03	1.40
Ground-water withdrawals, saline	MGD	0	0
Surface-water withdrawals, saline	MGD	0	0
Total withdrawals, saline	MGD	0	0
Total withdrawals, total	MGD	13.03	1.40
Water deliveries, public use and losses	MGD	6.17	0.66
Water deliveries, total deliveries	MGD	6.86	0.74
Per-capita use	GPD	129.06	
Number of facilities	--	22	2.37

An expanded analysis can be obtained from USGS for any accounting unit and includes water uses for the following items, where the total Upper-Ohio Shade HUC water withdrawal budget is shown below.

- Public Supply (13.03 MGD)
- Commercial (0.05 MGD, 1.18 MGD with deliveries)
- Domestic (1.14 MGD)
- Industrial (9.63 MGD)
- Fossil-fuel Thermoelectric Power (2239 MGD)
- Geothermal Thermoelectric Power (0)
- Nuclear Thermoelectric Power (0)
- Mining Use (2.38 MGD)
- Total Livestock (0.45 MGD)
- Irrigation (0.01 MGD)
- Hydroelectric Power (7800 MGD – instream use)
- Wastewater Treatment (16.2 MGD)
- Reservoir Evaporation (52,290 acre-feet)

Meigs Mine No. 31 Outfall-001 represents most of SOCCO's operations that divert groundwater into the Leading Creek Watershed. Total flow from the Meigs Mine No. 31 Outfall-001 averaged 4 MGD in 1997, indicating the above analysis by USGS is not complete. Mentioned below, the SOCCO discharges all are currently assigned to the USGS HUC=05030204, instead of HUC=05030202, which may account for the error. Average annual flows and average Total Dissolved Solids (TDS) for Meigs Mine No. 31 are described below.

2.10.4.2 Leading Creek Watershed Water Supply Data

In Leading Creek, all potable water uses are actually supplied by groundwater use, via either public or private wells. This includes SOCCO potable water needs as well. Domestic per capita use is estimated by USGS in this area to be 44.3 gallons per day (GPD) for public supplies, and 78 GPD self supplied rates. Self-supplied use rates are likely incorporating some farming uses. A total of 1.14 MGD is supplied from individual groundwater withdrawals within the Leading Creek Watershed. For Leading Creek, a total of 0.07 MGD of freshwater supplies may actually be used for consumptive purposes.

The Leading Creek Conservancy District (LCCD) currently provides public water supplies delivered to residents within the Towns of Langsville, Harrisonville, and Rutland Ohio. The LCCD at this time is operating 4 to 5 wells located behind the "Guiding School" in Cheshire, Ohio, south and outside of the Leading Creek Watershed. The District pumps from the Cheshire location to a water tank just west of the watershed, for apparent access to the Rt. 124 corridor area. The Town of Rutland, Ohio distribution system appears to also require periodic routine monitoring due to its size. The Town of Dexter, Ohio has initiated public works development, and soon will be hooked up to Leading Creek Conservancy District water supplies.

The Town of Albany is served by LEAX water, a commercial supplier. Their water supply is currently drawn from the Snowden Lake reservoir in Athens County. An aboveground water tank is shown in the Town of Albany on the USGS 7.5-minute quadrangle. This tank may serve as storage for the LEAX system, though this was not verified. Other residents in the watershed are self-supplied. Residential and agricultural water uses in this area may depend on dug or drilled wells likely drawing from aquifers in sandstone in the uplands, and sand and gravel layers on valley floors (Lucht *et al*, 1985). USGS and NRCS (modern SCS) report some private self-supplied uses of surface water for consumptive purposes may be derived from ponds, cisterns, and springs. None are known to specifically occur within the watershed boundary, though these collection techniques may be occasionally practiced in Leading Creek.

2.10.5 Natural Resources Extraction

As one estimate in Leading Creek watershed, there are 813 ha of abandoned surface mines, 1324 ha of abandoned underground coal mines and 4 acres of mine refuse (USDA, 1985). These figures for abandoned surface and underground mining were found to be underestimated when compared to coverages on USGS 7.5-minute quadrangle series and other geodata gathered through this study. As discussed above, the Pinegrove soil series indicating primarily ASML activities was not available in digital form.

2.10.5.1 Underground Mining

Excluding historic and current SOCCO operations, most underground mining in the Leading Creek vicinity was conducted prior to 1910, with some abandoned mines created in the 1930's and after WWII. Post WWI and WWII underground mining was tied in some cases to strip mining activities that became increasingly active from the 1930's onward. Based on records at ODNR, coal was reportedly shipped from Pomeroy, Ohio as early as 1833. Though minimal in comparison to later activities, there was likely some localized underground mining in Leading Creek prior to the Civil War. Most abandoned underground mining (AUML) in Leading Creek though occurred in the latter portions of the 19th century. A total of 1609 hectare of AUML was calculated for Leading Creek, representing 4.1% by area of the watershed. This figure excludes all SOCCO operations. Some of the entrances for the 124 separate abandoned underground mine works in Leading Creek can still be found today but only a handful are actually marked on USGS 7.5-minute quadrangles.

Minor amounts of coal were likely extracted from the lower Conemaugh Group. Below this formation, the SOCCO deep mines work the Clarion #4A coal seam. This seam is part of the Allegheny Formation, a 100m thick assembly of marine sandstone, siltstone, shale, limestone, and multiple coal units with estimated age of 330 million years. Due to its marine origins, higher sulfur and alkalinity might be expected in the Allegheny Formation compared to surface lithologies (Caruccio *et al*, 1974; Younger, 1996), but on average this did not hold true for coal deposits in the nearby vicinity of Leading Creek (USGS, 1998). Significant deep long wall underground mining in the basin is active today and is believed to have started sometime in the late 1930's.

According to the Meigs County soil survey (Gillmore *et al*, 1991), present coal mining today is restricted to deep (>100m below land surface) underground long-wall mining activities. These activities are conducted exclusively by SOCCO. Current SOCCO operations are still conducted at two active mines in the watershed, at Meigs Mine No. 2 and Meigs Mine No. 31. The latter mine operation was originally combined to designate previous mining operations at Mine No.1 and Mine No. 3.

2.10.5.2 Surface Contour Mining

Most strip mining in Leading Creek occurred after WWII based on interviews with watershed residents that indicated a significant increase in these activities in the 1950's. Based on aerial photo revision dates, 80% or 1379 ha of strip mines in Leading Creek were created by 1961, and 20% or 353 ha between 1961 and 1978. Little surface mining occurred after this period due to the onset and burdens of increased financial and environmental protection imposed by the Surface Mining and Reclamation Act of 1977. Excluding Thomas Fork where ASML was not digitized, only 4.2% by area overlap of ASML and AUML coverages was found in subsheds with AUML>0%. AUML was centered below ridges and ASML followed outer edge contours.

The typical case was that strip mining occurred after underground mining unless an abandoned highwall created by strip mining was further excavated using underground mining techniques. Near-surface underground and strip mining in Leading Creek were dominated by the excavation of Redstone #8A coal referred to as the Pomeroy #8A in Ohio, named after the Town of Pomeroy on the western shore of the Ohio River. The Redstone #8A formation is part of the Monongahela Bedrock Group, an estimated 90m thick sequence of non-marine sandstones, siltstones, shales, and coal with estimated age of 295 million years. The Monongahela Group would be expected to offer overall low to medium carbonate buffering capacity (USEPA, 1980).

According to one resident who was born in the watershed and graduated from the local high school in 1956, significant surface mining began in the early 1950's. Based on recollections of other residents, these time frames appear relatively accurate and coincide with stories of occasional fish kills, landslides, and other noted impacts rising during the later half of the century. Of greatest concerns typically noted were overburden sediment deposited from these operations in streambeds and AMD derived from oxidation of pyrite, a mineral iron-sulfide ore that occurs in and around coal seams (Nordstrom, 1982). The actual source of AMD, ASML versus AUML, is not well reconciled to date. Answering this question coincidentally became a major aspect of work investigated here.

2.10.5.3 Degree of Impact of Abandoned Mined Lands

Large-scale pyrite oxidation begins by exposure to oxygen through disturbance of the soil/rock profile. Once initiated, complete oxidation is nearly impossible to stop (Snoeyink *et al*, 1980). Oxidation is greatly facilitated in unsaturated environs by the efficient capacity of ferrooxidans, or "iron eating bacteria" that mediate pyretic oxidation reactions. The pyretic oxidation reactions have two major stages of acid generation, transforming reduced iron (as pyrite) to a precipitate oxyhydroxide that settles-out as fine silt coatings on coarser sediment instream (Bigham *et al*, 1996; Nordstrom, 1982). Consensus built upon today indicates that the kinetics of the iron-bacteria at low pH (<6 s.u.) dominate much slower chemical oxidation of pyrite (Williamson *et al*, 1994). With ferrooxidans, rates of oxidation are over 10^6 time greater than in sterilized water (Evangelou, 1995; Singer *et al*, 1970; Snoeyink *et al*, 1980; Williamson *et al*, 1994).

According to previous studies of the watershed and surrounding areas in southeastern Ohio, erosion rates up to 200 tons per acre per year were assessed over a large footprint of mining activities in Meigs, Athens, and Gallia counties (USDA, 1985). The associated survey of 30 southeastern Ohio counties ranked the Leading Creek watershed highest in sediment damage and acreage of deposition associated with AML. Total erosion and erosion rate were also ranked second highest, and loss of useful land was ranked third highest (USDA, 1985). Reportedly, similar quotes from Ohio EPA officials have also been made at local public meetings in recent years, where Ohio EPA has continued to note that Leading Creek appears to be one of the worst sediment-laden watersheds in the State of Ohio.

2.10.5.4 Rural Abandoned Mines Program (USDA-RAMP)

In discussions with Byron Thompson, currently the National Coordinator for USDA's Rural Abandoned Mines Program (RAMP), he recounted some of the successes and failures of earlier efforts to reclaim AML in Leading Creek and southeastern Ohio. AML reclamation in southeastern Ohio was begun in the late 1970's, after SMCRA regulation in 1978. To their credit, the Soil Conservation Service (SCS) at the time attempted to establish a watershed management based approach. This entailed mapping out the problem and establishing a process of prioritization, much the same job faced today (USEPA, 1992, 1998). The SCS arrived at the current system still in use today at that agency where they evaluate up to 6 levels of categorization of AML/AMD impacts.

It is paramount to understand several key issues that related to that earlier work by SCS (now USDA-NRCS or National Resource Conservation Service). AML related Office of Surface Mining (OSM) work and USDA-NRCS work is ongoing with a recent reprioritization in policy at the end of the 1990's towards addressing AMD water quality issues. Emphasis also is now placed on increased coordination within other State and Federal agency programs.

The following comments are summarized from interview notes taken with Mr. Thompson (NRCS, 1998) and Mr. Joseph Bolin. Mr. Bolin is a local RAMP community leader/organizer on AMD/AML issues in the Leading Creek Watershed.

- ❖ The highest priority ratings (Level 1 and 2) were addressed first, representing most of the work that has been completed to date in southeastern Ohio. In their approach, NRCS set level 1 and 2 based on physical hazard mitigation, e.g. flooding issues basically. Work entailed a great deal of tree plantings and vegetative cover restabilization efforts.

- ❖ NRCS identified two basins early on with severe problems, Leading Creek Watershed and Duck Creek Watershed in Noble County near Marietta, Ohio. These watersheds were given generally highest priority. Other watersheds were also eventually selected and slated for significant reclamation activities.

- ❖ NRCS's survey techniques tended to concentrate on data collection and address of subsheds with observable road flooding problems. The process was apparently not methodical in addressing all subsheds without roads, or those with roads but without severe flooding.

- ❖ Inventories maintained by NRCS may be incomplete due to the above aspects. Mr. Thompson, one of the developers of the original prioritization process and list, commented that he would not be surprised to find disagreement between original NRCS inventories and other more methodical inventories completed today, for instance using quantitative spatial analysis as used here.

- ❖ Reclamation of the Harrisonville area began around 1978. At the time, local stream conditions on Little Leading Creek were extremely degraded in this area. Sediment was so heavy that it filled the entire channel at some cross-sections, reaching bridgedeck levels. A picture was taken at one site confirming this fact and another taken just last year (~15 years later) at the same location. The pictures showed that the channel had recovered where now 6-7 feet of headspace between the bed and bridgedeck could be observed.

- ❖ According to Mr. Thompson, NRCS retains the understanding that earlier efforts in the watershed by and large arrested sand deposition, where not all areas may have been addressed.

- ❖ Based on the efforts undertaken, NRCS believed that significant improvements were also being realized for water quality, where reclamation of physical issues concurrently addressed some of the AMD problems.

- ❖ If it were to be done again, AMD issues would have been given higher priority earlier on, where it is now believed that AMD has not been adequately addressed to date.

- ❖ The current focus of NRCS work is increased prioritization for address of AMD water quality related problems in affected watersheds.

As of 1997, according to Mr. Joe Bolin, Area 5 Director, Ohio Federation of Soil and Water Conservation Districts:

- ➢ In Leading Creek there was one ongoing AML project designed and scheduled for bids; ~$500,000 on 47 acres; Titus Road Project on Titus Run (i.e. station TS15).

- ➢ One organizational group, the Rutland Watershed Flood Committee had secured $1,048,000 in hazard mitigation grants (HMGIP). Though specific lists of projects were never obtained, the projects appeared to be sediment-maintenance oriented.

- ➢ Organized through NRCS, another $1,800,000 may have also been secured for other potential reclamation/restoration work in the watershed. Though specific lists of projects could not be obtained, the projects were primarily characterized again as sediment-maintenance oriented.

2.10.5.5 Oil and Gas Well Production

Due to the energy crisis of the 1970's and increased economical viability, a preponderance of well heads, mechanical ants, oil tanks, and plastic brine tanks now dot the landscape of Leading Creek and surrounding areas. Oil and gas wellheads may be found typically in portions of the lower and eastern parts of the watershed. Production levels are greatly decreased today, but appear to remain active for the more productive wells and some smaller wells.

Based on discussions with local oil operations personnel and landowners, relatively relaxed environmental control and regulation characterized oil and methane gas production up to the early 1980's. The recovery of oil, natural gases, and naturally occurring brines (saltwater) are common at most petroleum extraction wellheads. The oil, vapor gases, and brine solutions are typically gathered at the surface, and then separated using a condenser and various mechanical devices and techniques. Land-use coverage on USGS 7.5-minute quadrangle series maps for gas and oil wells may be somewhat misleading. Reportedly natural gas (as vapor) extracted in these areas are likely composed of several dozen hydrocarbon gases other than pure methane.

Previously, contaminated brines were reportedly discarded commonly in pits nearby the well heads as discharges to groundwater and in some cases more directly as surface run-off. During run-off events, pulses of material discharged from depression storage were probably common, along with discharge of impacted groundwater at nearby stream banks.

More recently, USEPA and the State of Ohio now require active containerization and management of brine wastes as well as addressing significant oil spills. The added over-site and management of wellhead wastes appears to have abated most observable impacts from these operations. Several collection systems do reside within the floodplain. This likely still results in some, but infrequent impacts, to aquatic ecology due to run-off from small spills and overflow, leaking lines, and flooding. Many transfer lines from well heads to holding tanks cross-stream channels, where local impacts may occur.

2.10.6 Silviculture – Forestry Management

Viewed here as a separate resource extraction activity, forestry management operations are characterized in Leading Creek by two basic groups of operators. One of the operations is the SOCCO/AEP forest-tract land management group, formally organized in 1981. The second group is the smaller local independent operators who supply wood materials to a local Kraft paper mill owned by Mead Corporation in Chillicothe, Ohio. Mead itself does not actively manage or harvest woodlands but simply purchases raw supplies. Up until mid 1980's Mead utilized primarily chipper wood materials (chipped in the field). Due to demands on paper quality, they more recently have gone to rounding wood (4"-12" straight stems sent directly to the mill), again used directly for paper production at the plant.

The SOCCO/AEP land holding totals 18,900 acres of surface ownership with a total 7,700 acres held within Leading Creek. There are minor areas in the land tract held by the Ohio Power Company and Franklin Real Estate Company. The larger management tract is collectively referred to as the Wellston Coal Field/Coal Conveyor/Gavin-Plant lands. SOCCO/AEP operations use a selected harvest management approach that reduces non-point source pollution and run-off.

Since 1987, SOCCO/AEP has followed the standardized best management practices (BMPs) for silviculture published by the Non-Point Source (NPS) Pollution Technical Advisory Committee (ODNR, 1992). A pro-active forest management plan has been in place since 1982, after completion of the tract inventory. Approximately 1400 acres of natural hardwood stands have been selectively harvested in Leading Creek since adoption of the plan along with 183 acres of pine clear-cut to promote hardwood regeneration. This represents 1.7% of the watershed over roughly two decades.

Since 1985, another 381 acres of the managed land tract within Leading Creek have been planted to trees (330 acres of pine, 51 acres of bottomland hardwoods). The latter areas included marginal agricultural lands and reclaimed grasslands created by AML reclamation projects. All SOCCO/AEP timber sales follow the advisory committee's BMPs, which include among other aspects use of water diversion techniques (e.g. turnouts, dips, culverts, outlet dissipaters, water bars), use of filter strips, reseeding, and reforestation. No significant impacts to aquatic ecology would be expected to occur today from SOCCO/AEP BMP operations.

A second group of local timber operations are less organized, do not typically maintain active forest management plans (the program is voluntary), and in general may or may not employ all standard BMPs. Due to the relatively sparse nature of these operations, significant impacts may occur, but are likely localized to nearby tributaries. Log skidders can be seen in occasional use on small tracts along the central western and northwestern areas of the watershed.

2.10.7 Agriculture – Farming Practices

Trends show farming is on the decline in Leading Creek and in Meigs County, the State of Ohio, and the U.S. in general. The local county soil survey also reported a recent trend in conversion of floodplains of the Ohio River to industrial uses. This conversion would be primarily to sand and gravel extraction. Such operations are outside the Leading Creek Watershed boundary and appeared to occur more frequently to the north along the Hocking River. In addition to 20-30 acre lot conversions to residential land use, silviculture personnel working in Leading Creek also noted a growing trend of conversion of farmland to brushland with eventually progression to woodland.

The sale of fresh vegetables, livestock, and livestock products accounts for most of the annual farm income at farms in Meigs County (Gillmore *et al*, 1991; MCCC, 1996). Livestock dairy and beef operations occur in the basin where dairy operations generate 49% of livestock cash receipts (MCCC, 1996). Approximately 13% of land in Meigs County are considered prime farmland (Gillmore *et al*, 1991). The category "other crops", as listed by the Meigs County Chamber of Commerce (MCCC) generated 76.8% of cash crop receipts in 1988.

Other trends in Meigs County farming uses reported are given below (MCCC, 1996):

- Average farm size is 189 acres, 51% of farms are 50-179 acres, and 24.7% of farms are 180-499 acres.

- Between 1973 and 1988, the average farm size increased from 160 acres to 189 acres, peaking in 1987 at 192 acres. A total of 45.5% of farmers work outside the farm, where 29.9% do not.

- Between 1973 and 1988, the total number of farms decreased from 730 to 530. The sharpest decline occurred between 1976 and 1977 when 70 farms were lost.

- Total farmland decreased in the same time period from 120,000 acres to 100,000 acres, representing a 17% decline.

Poor farming techniques and practices may be on the rise in some locals, where BMPs can be implemented less frequently due to economical stresses placed on these operations. Channelization and encroachment of stream corridors are common problems in Leading Creek (USDA, 1998).

2.10.8 Regulated Point Source Discharges (NPDES)

To provide a perspective of how nonpoint source pollution associated with AG and AML might relate to ecology, knowledge of point source pollution is of course also important to provide a level of control or reference point. The National Pollutant Discharge Elimination (permit) System was reviewed for point source discharges in to Leading Creek. Information was made available through electronic records maintained by SOCCO and USEPA; the latter was obtained via the Internet. Information used in the evaluation was also accessed, analyzed, and developed from NPDES permit compliance monitoring data using USEPA's Mor Software (both data and computer code). Table 2.27 lists various NPDES permits held within Leading Creek.

2.10.8.1 Miscellaneous NPDES Discharges

The limestone quarry operations just below Albany, Ohio are currently closed. The associated effluents appear to be derived solely from management of run-off at this time. The quarry adds some alkalinity buffering capacity to upstream waters. The quarry effluents likely benefit lower-mainstem areas impacted by AMD.

The Leading Creek Conservancy District (LCCD) has a permit allowing discharge from its plant operations. The discharge is permitted into Little Parker Run just above its confluence with Parker Run. The permit for the water supply plant limits TSS, suspended iron and manganese, and pH. Both the WTP and quarry discharges are expected to have minimal impact on flow, pollution, and aquatic ecology in downstream areas.

2.10.8.2 Rutland Sanitary Sewerage Wastewater Treatment Plant

According to Dale Hart, Racine-STP System Operator who oversees the Rutland Sewage Treatment Plant (STP):

Table 2.27 – National Pollution Discharge Elimination System (NPDES) Point Source Summary for Leading Creek

Facility	NPDES ID	Outfall Tributary	Lat	Lon	SIC	Description	Monitoring Parameters
DIAMOND STONE QUARRIES Tributary T58	OH0091162	Below Town of Albany (T58) - See Station LCS1	Location Estimated	See Map	3295	MINERALS GROUND OR TREATED	pH, TSS, Flowrate
SOUTHERN OHIO COAL CO (SOCCO) Meigs Mine No. 2	OH0022837	Unnamed headwater tributary of Ogden Run (T41) - See Station TS6	39 07 29	-82 17 56	1221	BITUMINOUS COAL AND LIGNITE - DEEP LONG-WALL UNDERGROUND MINING	Temp, Flowrate, Color, pH, TSS, NH3 as N, Total Fe, Total Mn, Odor, Turbidity, Coliform, Rainfall, Chlorine (TRC), CBOD5, 20C
SOUTHERN OHIO COAL CO (SOCCO) Meigs Mine No 31	OH0022829	Unnamed headwater tributary of Parker Run (T28) - See Station TS11	39 04 15	-82 14 15	1221	BITUMINOUS COAL AND LIGNITE - DEEP LONG-WALL UNDERGROUND MINING	Temp., Flowrate, Color, DO, pH, TSS, Oil & Grease, NH3 as N, Total Fe, Total Mn, Odor, Turbidity, Coliform, Rainfall, Chlorine (TRC), CBOD5, 20C
LEADING CREEK CONSERVANCY DISTRICT	OH0099279	Unnamed tributary of Little Parker Run (T28) See Station TS11	Location Estimated	See Map	4941	WATER SUPPLY	pH, TSS, Flowrate, Suspended Fe, Suspended Mn
RUTLAND STP	OH0050130	Little Leading Creek (T11) See Station TS14	Location Estimated	See Map	4952	SEWERAGE SYSTEMS	Temp., Flowrate, Color, DO, pH, TSS, NH3 as N, Odor, Turbidity, Rainfall, Chlorine (TRC), CBOD5, 20C , E.COLI, MTEC-MF, Bottom Deposits: Cr, Cd, Cu, Hg, Ni, Pb, Zn, %Solids %Volatile Solids

- Average monthly flow from the STP is 0.5 to 0.75 million gallons or 0.02 MGD,
- Activated Sludge, Dual Tanks - Forced Air; Residential Package Plant System
- Effluent CBODs (carbonaceous biological oxygen demand) average approximately 2-3 mg/l, and discharges never go above 3 or 4 mg/l NH_3 (DO driven standard),
- Mean hydraulic residence time is 3 days.

Minor domestic SOCCO flows are derived from typical sanitary plant effluents at Mine No 31 (35,000 GPD design - 4100 GPD actual) and Mine No. 2 (30,000 GPD design – 10,900 GPD actual). The plants also utilize aerobic digestion with rapid sand filtration polishing, chlorination, and dechlorination.

2.10.8.3 Coal Mining/Preparation Plant Process Wastewater Discharges

AMD treatment plant effluents (e.g. treated wastewater) from dewatering each mine operation and coal preparation plant effluents from Mine No. 31 are fed to a slurry impoundment at Mine No. 31. The slurry impoundment and surface run-off at Mine No. 31 are conveyed to the freshwater (raw water) reservoir at Mine No. 31 area, which also provides final polishing and equalization/storage control.

The reservoir discharges vary based on groundwater production by the two mines. Groundwater production in the mine areas can be quickly influenced by rain events. The freshwater reservoir supplies most non-consumptive water uses at both plants. Recycled make-up water for process is pumped from the Mine No. 31 reservoir to a 100,000-gallon redwood tank at the No 2. Plant, for interim storage. In response to higher run-off events, Mine No. 2 discharges water overflow from the AMD holding pond along with direct facility surface run-off. Flow occurs through a gravity thickener unit and settling pond used to remove solids. Thickener effluent is directed through two ponds with relatively low equalization and eventually discharged to a tributary of Ogden Run after physical treatment.

Total average flows and Total Dissolved Solids (TDS) for Meigs Mine No 31 Outfall-001 are given in Table 2.28. Most process wastewaters from Meigs Mine No 2 are discharged into Parker Run via the Mine No. 31 reservoir. Trends in Table 2.28 indicate a general increase in flowrate and load rate and a slight decrease in concentrations since 1990.

Table 2.28 - SOCCO Mine No. 31 Reservoir Effluent Outfall-001 Discharge to Parker Run

Year	Average Annual Flow (MGD)	Average Total Dissolved Solids (g/L)	Average Total Dissolved Solids (tons/day)
1990	0.46	5.4	10.3
1991	0.60	5.7	14.2
1992	0.98	6.5	26.6
1993	1.3	7.2	39.8
1994	2.5	6.1	63.6
1995	1.9	5.2	40.3
1996	3.8	4.5	71.5
1997	4.0	4.4	73.8

2.10.8.4 Point Source Discharge Summary – Leading Creek

Summarizing the NPDES point source (PS) discharges to Leading Creek in Table 2.27, total flows associated with the following operations appear negligible with respect to total flows monitored at various mainstem and tributary stations in this study:

- ❖ Rutland Sewage Treatment Plant (STP),
- ❖ Conservancy District Water Treatment Plant (WTP),
- ❖ Limestone quarry operations, and
- ❖ All SOCCO sanitary STP facilities

The sewage treatment plant operations are not expected to provide significant loading of nutrients or metals given the small populations they serve (Metcalf & Eddy, Inc., 1991). The latter assumes that reasonable treatment performances are attained for CBOD at package plants. The limestone quarry operations are currently closed with flows derived from post-closure management of baseflow and stormwater run-off.

Discharging to a tributary of Parker Run, SOCCO Meigs Mine No. 31 Outfall-001 represents most of SOCCO's operations that divert groundwater into the Leading Creek stream system. Total flow from the Meigs Mine No. 31 Outfall-001 averaged 4 MGD in 1997, with elevated TDS at 4400 mg/l. The TDS salts are composed primarily of sulfate and sodium representing by-products of treatment of AMD by pH neutralization with sodium hydroxide. Significant effluent flows monitored at the Mine No. 2 represent mostly thickener operation effluents associated with elevated groundwater production after storms. On average, Mine No. 2 discharges less frequently, and has lower salt content than Mine No. 31 effluent (e.g. Mine No. 2: typical flow ~ 0.2 to 2 MGD when flowing, TDS up to 1500 to 3000 mg/L).

At both Meigs Mines No. 31 and No 2, pH is alkaline, typically at about 8.0 standard units. Elevated chlorides may also be present (200-400 mg/L), and both iron and manganese are typically low, between 100 and 1000 ug/L. According to these levels for coal mine plant effluents, trace heavy metals should be below established threshold concerns (USEPA, 1976a). Increased effluent flows due to rain events will lag storm-flows to some degree. Both mines manage stormwater run-off through a series of settlement ponds, and appear to maintain relatively low TSS in discharges.

2.11 HISTORICAL IMPACT ALONG THE OHIO RIVER AND LEADING CREEK

2.11.1 Sub-Regional Water Quality Assessments

The following material was taken from ORSANCO's and provides a succinct summary of sub-regional water quality trends on the Upper Ohio River and tributaries (CVC, 1998):

> *"Water quality in the Ohio declined after about 1810, when people began moving into the Pittsburgh area in large numbers. The River's quality declined until about the 1940s because of industrial pollution, raw sewage, mine drainage and*

other problems. In the 1940s, mine drainage from active and unregulated coal mining caused the pH (a measure of acidity) to be as low as 4.0 in the upper Ohio River.

Untreated sewage resulted in monthly total coliform bacteria counts exceeding 20,000 per 100 ml of water, according to information compiled by the Pennsylvania Fish & Boat Commission. After outbreaks of waterborne diseases in the 1930s and 1940s, the Ohio River Valley Water Sanitation Commission (ORSANCO) was formed in 1948. ORSANCO promoted the construction of sewage treatment systems in the 1950s and 1960s.

The Clean Water Act began to require active industry to clean up discharges, and the upper Ohio River made a striking recovery by the mid-1970s. Fish populations expanded dramatically since then, and include such pollution sensitive species as walleye and sauger. However, problems with contaminated sediments and fish flesh of bottom feeders have persisted. Carp and Channel Catfish, which feed in sediments, have high levels of PCB and Chlordane. Other species of gamefish and panfish do not have this contamination, according to the Fish & Boat Commission. Species of sportfish in the River near Pittsburgh include bass, freshwater drum, crappie, walleye, sauger, bluegill, carp, tiger muskellunge and suckers.

Today's water quality problems in the Ohio River and its major tributaries are generally the result of small contributions of pollution from many sources, rather than large discharges by a relative few sources as was the case 20 years ago. Studies to determine the specific contributions of the numerous sources, and the effects of various control options, are highly complex and generally beyond the financial capabilities of the Commission. Such studies are also generally too costly for any single discharger to conduct. Many regulatory decisions must therefore be made without adequate supporting data.

ORSANCO also reported the following population and water use data (CVC, 1998):

- ✓ The Ohio River is a drinking water source for 3 million people,
- ✓ 10 % of the US population (~25 million) lives within the Ohio River Basin,
- ✓ Urban run-off, agricultural activities and AML are the major causes of water pollution in the Ohio River.

The Ohio River Commission is also attempting to facilitate cooperative efforts supporting data collection for regulatory decisions. The Commission established a program under which interested parties may contribute to support applied research on Ohio River water quality (CVC, 1998). The Commission’s steering committee recommends projects for approval and funding. The first project funded was in 1995. The project focused on development of a biological database and information management system. When completed, the system will serve as a repository for data collected by industries, universities, and government agencies (CVC, 1998).

2.11.2 Use-Attainment Analysis: OEPA Watershed ID Group 29

The following ecological evaluations were completed by Ohio EPA and were studied here as part of the initial risk assessment. The 1996 Ohio Water Resource Inventory (305b Report) materials were made available through Ohio EPA's Explore Your Watershed website.

(http://chagrin.epa.ohio, http://chagrin.epa.ohio.gov/watershed/npsa/npsa29.htm).

> **According to Ohio EPA:** *Use attainment is another way of describing whether or not a stream is meeting Ohio's water quality standards. Ohio EPA has assigned a use designation- or a specific set of water quality standards- to most major streams and rivers throughout the state by dividing each stream into segments and assigning each segment a specific use designation. However, not all stream segments are designated and not all designated segments have been monitored.*

Based on the Ohio EPA's designated watershed description for Group 29-Lower Raccoon Creek on the "Find Your Watershed" website, the accompanying use attainment summary Tables 2.29 and 2.30 were developed. Data was reportedly based on the 1996 Ohio Water Resource Inventory (305b Report). See Figure 2.1 and Figure 2.2 above for a description of the associated watershed ID system. The Leading Creek Watershed is included within Group 29.

Table 2.29 - Stream Miles Assessed For Use-Attainment – OEPA Group 29
(Source: Ohio EPA, Find Your Watershed Database – October 1998)

Percentage of Assessed Stream Miles Attaining	37.3%
Percentage of Assessed Stream Miles Not Attaining or Partially Attaining	62.7%
Percentage of Total Stream Miles Assessed	32.6%

Table 2.30 - Aquatic Life Use-Attainment Designation Summary – Group 29
(Source: Ohio EPA, Find Your Watershed Database – October 1998)

Aquatic Life Use Designation	***Use Attainment Status***					
	Total Stream Miles Designated	**Miles Assessed**	**Miles Fully Attain.**	**Miles Threatened**	**Miles Partially Attain.**	**Miles Not Attain.**
Exceptional Warmwater Habitat	0	0	0	0	0	0
Warmwater Habitat	181.9	68.4	19.6	4.6	10.2	34
% Warmwater Habitat	100	38%	29%	7%	15%	50%
Modified Warmwater Habitat	0	0	0	0	0	0
Limited Resource Water	21.5	2.4	0	0	0	2.4
Coldwater Habitat	0	0	0	0	0	0
All	228.3	74.3	23.1	4.6	10.2	36.4
Not Designated	24.9	3.5	3.5	0	0	0

The Leading Creek Watershed is included in OEPA Group 29.

For Table 2.29 the following definitions

- **All -** represents total designated and not designated (or undesignated) stream miles. However, this total does not include undesignated/intermittent streams or designated but not monitored streams that are shown on the group map.

- **Not Designated -** represents the number of stream miles that have not been assigned an aquatic life use designation. However, undesignated stream segments are still subject to Ohio's narrative water quality standards.

2.11.2.1 Causes of Impairment

According to Ohio EPA, the causes of impairments listed in Figure 2.5 are for those stream segments that have been assessed and reported in the 1996 Ohio Water Resource Inventory (305(b) report). OEPA notes that a stream mile may be impaired by more than one cause, and the total impaired miles shown may not equal the total number of impaired miles in the hydrologic unit (as displayed in the use attainment summary Table 2.30 and in the source 305b segment data tables).

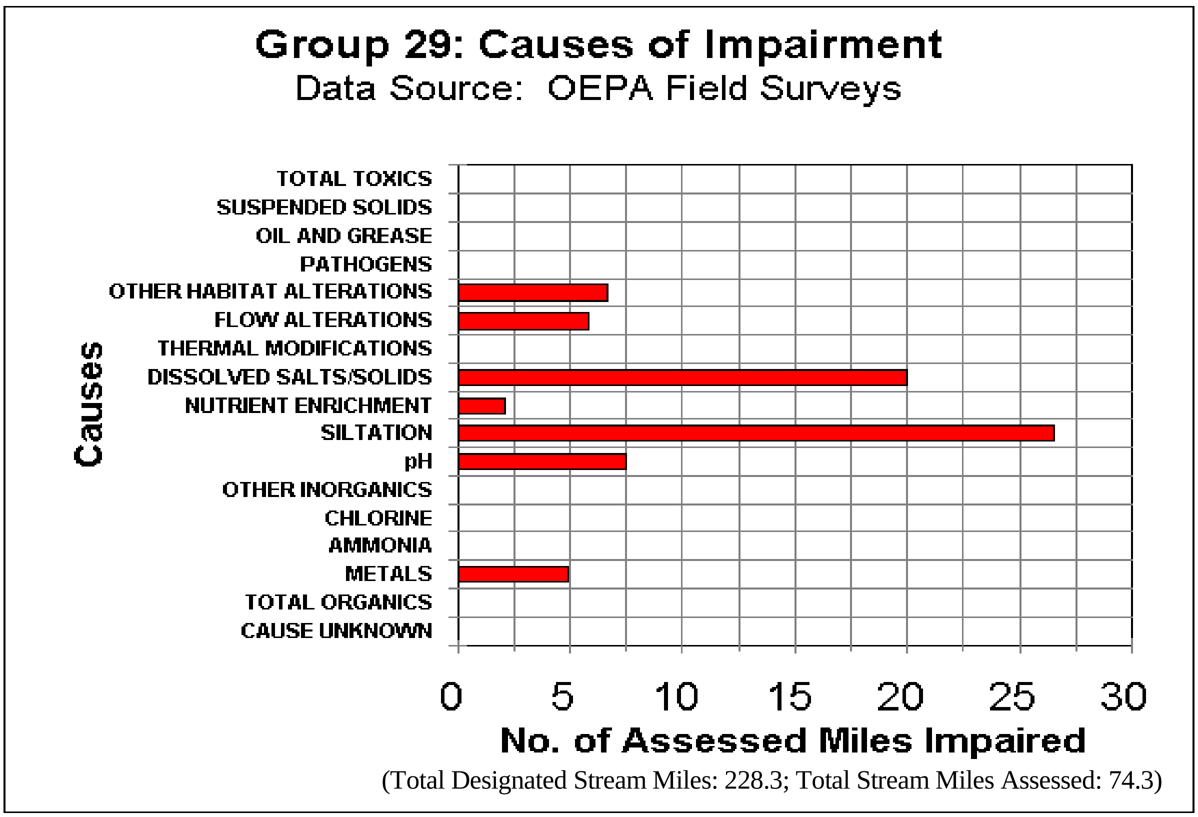


Figure 2.5 - Ohio EPA Causes of Impairment Summary – Group 29

2.11.3 Ohio EPA Biological and Water Quality Studies - Reference Watersheds

As part of the research conducted here, two reference watersheds, Symmes Creek and Federal Creek, were also monitored. These creeks along with all of the southeastern Ohio streams together reside in the Western Allegheny Plateau Ecoregion, one of four ecoregions covering the state (OEPA, 1991a, 1991b; USEPA, 1988). The Symmes Creek, Federal Creek, and Leading Creek watersheds have been previously evaluated by Ohio EPA in on-going biological and water quality studies. The studies parallel to degree the reach specific surveys of the hydrologic unit “Groups” identified previously, though single studies intersect both Ohio’s and USGS’s HUC watershed identification systems.

2.11.3.1 Federal Creek Watershed

Federal Creek, a primary tributary of the Hocking River, spans Athens and Morgan Counties and is monitored within Ohio EPA’s Biological and Water Quality Study of the Hocking River Mainstem and Selected Tributaries (OEPA, 1991a). Federal Creek’s mainstem drains 37,450 ha and is 38 km in length (23.8 miles). The subwatershed area monitored for this study is 6790 ha comprising the northwestern headwaters, shown in Figure 2.3. The average energy line fall of Federal Creek mainstem was noted by OEPA to be 19.1 feet/mile or 0.36% slope. Federal Creek was also generically noted by OEPA to have “in-place contaminants” (OEPA, 1991a) and depending on reach may be subject to severe AMD and sedimentation from ASML.

2.11.3.1.1 1991 Hocking River Study

- OEPA again reassigned the EWH (Excellent Warmwater Habitat) designation given to the upper section of Federal Creek (above Sharp’s Fork). Previously EWH attainment status was also assigned in the 1978 Ohio WQS (Water Quality Standards). These evaluations were based on observation of adequate habitat and good water quality.

- Declining indications in water quality were noted at Federal Creek in 1996. Though reconnaissance results were good on earlier visits in fall of 1995 and spring of 1996, conditions appeared to deteriorate over the summer and into fall of 1996.

- A small area of AML is located just upstream from the site above Amesville, and was previously thought to be innocuous. The problems observed at Federal Creek included typical diffuse AMD impact signals. Increased specific conductivity and increased mortality and widespread degradation to community structure over the course of monitoring.

- AML impacts in Sharp’s Fork and McDougall’s Branch can influence the lower portions of Federal Creek resulting in Limited Warmwater Habitat (LWH) designations, though EWH status was most recently observed.

- To better assess any potential changes in use-attainment character within upper portions of Federal Creek in 1996 and 1997, this study also incorporated additional reference data from Symmes Creek, discussed below.

2.11.3.2 Symmes Creek Watershed

Symmes and Leading Creek are both monitored within Ohio EPA's Biological and Water Quality Study of the Southeast Ohio River Tributaries (OEPA, 1991b). Symmes Creek, like Leading Creek is a primary tributary of the Ohio River that confluences at Ohio River mile 308.7 in Chesapeake, Ohio, (Lawrence County), across the river from Huntington, WVA. The mainstem of Symmes Creek is 70 miles or 113 km in length, and drains an area of 92,000 hectares (CVC, 1998). The upper and lower mainstem reference site sampling locations studied here each drain 34,800 ha and 89,700 ha, respectively. The average energy line fall of Symmes Creek mainstem was noted by OEPA to be 3.4 feet/mile or 0.06% slope. Leading Creek's average fall was 8.4 feet/mile or 0.15% slope (OEPA, 1991b).

2.11.3.2.1 1991 Southeast Ohio River Tributaries Study

Ohio EPA's survey evaluated 29 stream segments on 18 streams in the subbasin which also encompasses the Leading Creek Watershed. Critical conclusions of that study are recounted below (OEPA, 1991b):

- ❖ Based on final OEPA recommendations, attainment status on 24 reaches (where fish and macroinvertebrate were both evaluated) would be:
  - FULL Attainment = 12 reaches
  - PARTIAL Attainment = 8 reaches
  - NON Attainment = 4
  - Cause for non-attainment was most typically attributed to AML sedimentation, water quality related impacts from AMD, and PS pollution from sanitary wastewater treatment STPs.
- ❖ Over all use-attainment was rated at both Leading Creek and Symmes Creek stations included in the study as meeting WWH or warmwater habitat use-attainment criteria.
- ❖ Lower Symmes Creek had marginally good (Miwb) to exceptional (IBI) fish community scores and exceptional macroinvertebrate habitat.
- ❖ Stations monitored in Campaign Creek, Leading Creek, and the Shade River all had fair fish community and exceptional macroinvertebrate structure.
- ❖ The exception was benthic macroinvertebrate on Leading Creek. This site at RKM 17.7 on the mainstem (equivalent to station LCS7) is upstream of the lower mining belt area, but downstream of the active mine discharges. The site supported only a good biological community with a notable near complete absence of mayflies.
- ❖ In general Leading Creek station LCS7 at RKM 17.7 maintained good habitat quality. Biodiversity may decline more severely in the lower reaches of Leading Creek with increased (AML and AMD) mine discharges. According to OEPA, upstream of the LCS7 study site, AML extent and impact declined significantly.

- Little Leading Creek was affected by AMD only in downstream reaches. Reportedly only minor impacts still exist due to AMD in areas upstream.

- Biological trends on the Leading Creek site included consistent, relatively flat trends in ICI scores from 1987 through 1991. Low 1988 data was attributed to early collection conducted on May 26th. ICI scores were 36 in 1991, 36 in 1990, 34 in 1989, 28 in 1988, and 32 in 1987.

2.11.4 USEPA Surf Your Watershed

The following national watershed database information was compiled by USEPA. It can be found at their website "Surf Your Watershed" by going to the Upper Ohio-Shade USGS Cataloging Unit: 5030202. As seen from the summary provided by USEPA, the Upper Ohio-Shade cataloging hydrologic unit (i.e. not OEPA Group 29) encompassing the Leading Creek Watershed is known for its lack of use-attainment, fish consumption advisories, and poor source water condition. According to USEPA much more information is apparently needed to better describe baseline conditions for water quality toxicants and conventional pollutants.

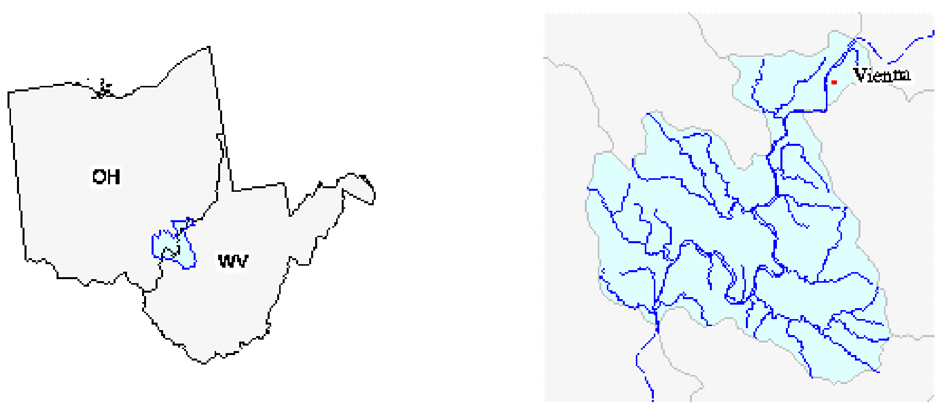


Figure 2.6 – Vicinity Map for USEPA Upper Ohio – Shade HUC 5030202

The overall Index of Watershed Indicators (IWI) in the Upper-Ohio Shade HUC was 5 of 6, where six represents the most serious water quality problems, nationally. The IWI index score is based on indicators of current condition and future vulnerability (USEPA).

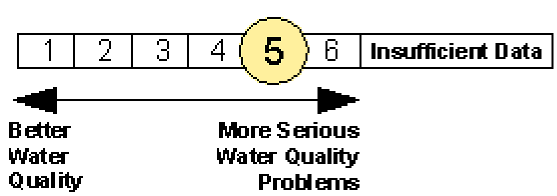
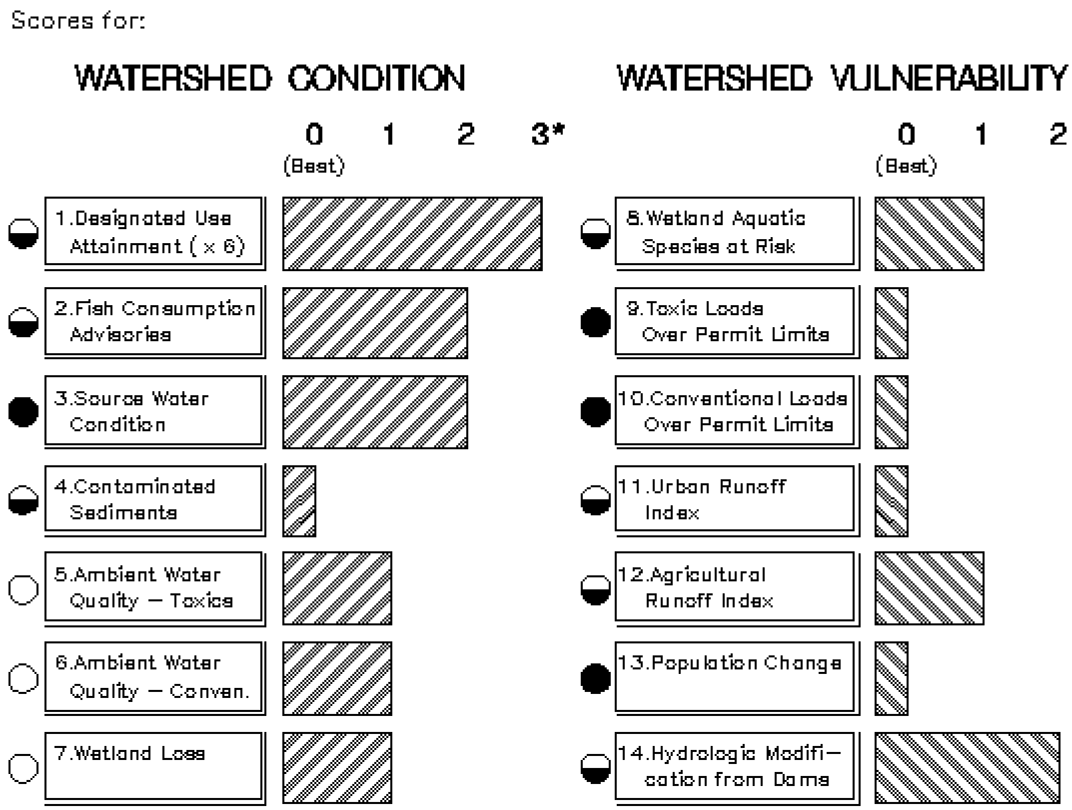


Figure 2.7 –Index of Watershed Indicator (IWI) for the Upper-Ohio Shade HUC 05030202

According to USEPA, the summary given above provides an indication of the quality of the national data set for this HUC. For any specific watershed, more or less data of adequate quality may be available. Underlying elements of watershed condition and vulnerability for the Upper-

Ohio Shade HUC 05030202 are given in Figure 2.8**.** Watershed condition and vulnerability in the Figures 2.7 and 2.8 represent baseline data as of October 1997.



Legend Key	Description
Full Moon:	Data Consistent. Sufficient Data Collected.
Half Moon:	Data Somewhat Consistent. Additional Data Needed.
Empty Circle:	Data Needs to be Much More Consistent; Much Additional Data Needed.

Legend Notes:

- *Watershed condition and vulnerability scores range from zero to three. The higher numbers reflect the more serious problems depicted by the data.*
- *Category 1 is the most important. It is the only category that can receive a score as high as three, and its score is then further weighted by a factor of six before it is used to compute the Overall Watershed Score*
- *In this case the IWI score was 5 where the worst-case 6 represents the most serious water quality problems nationally.*

Figure 2.8 –Watershed Condition and Vulnerability for Upper-Ohio Shade HUC 05030202

2.11.5 Acid-Sensitivity and Impacts on Biota

Revisiting USEPA's acid rain deposition study (USEPA, 1980) discussed for bedrock characteristics, USEPA found that contemporary studies of AMD at the time had similar conclusions regarding biota. USEPA concluded that related acid-base precipitation chemistry and toxicology was more complicated though when mining discharges were involved. This would obviously make extrapolation from AMD acidification to atmospheric deposition and associated stream acidification presumably more difficult. Due to its level of simplicity though, the perspective of atmospheric acid rain deposition does significantly inform our knowledge about impacts from AMD. It does so somewhat by separating effects of pH and ecological response from chemical flocs and heavy metals, and other presumably less impacting but possibly still important chemical traits of AMD.

The chemical effects noted by USEPA are also described in more detail later in AMD mechanisms. Added chemical effects from AMD are due to the presence of heavy loads of iron and aluminum, heavy metals, and chemical floc that can greatly alter instream conditions (Diz, 1997; Gray, 1996; Nordstrom, 1982). Specific results of the USEPA study included the following Tables 2.31, 2.33, and 2.33 reprinted here from the original study (USEPA, 1980). In developing the tables, USEPA first described effects likely to result from increasing acidification, Table 2.31. USEPA subsequently provided step-functions for damages that may be used in modeling ecosystem acidification, based upon an anticipated resultant instream pH after acidification. Table 2.32 addresses various levels of biologic organization. Table 2.33 addresses fish population effects due to acidification. USEPA's results are useful in discerning minimum impacts that might be attributed to AMD and its associated two stages of acid generation (Nordstrom, 1982).

Table 2.31 - Damages to Aquatic Biota Likely to Occur with Increasing Acidity;
(Adapted from USEPA, 1980)

Item	Effects Progressing Towards Decreased Biotic Integrity
1	Bacterial decomposition is reduced and fungi dominate saprotrophic communities. Organic debris accumulates rapidly.
2	The ciliate faunas are greatly induced.
3	Nutrient salts are taken-up by plants tolerant of low pH (mosses, filamentous algae) and by fungi. Thick mats of these materials may develop which inhibit sediment-to-water nutrient exchange and choke out other aquatic plants.
4	Phytoplankton species diversity, biomass, and production are reduced.
5	Zooplankton and benthic invertebrate species diversity and biomass are reduced. Remaining benthic fauna consists of tubificids and chironomus (midge) larvae in the sediments. Some tolerant species of stone flies and mayflies persist as does the alderfly. Air-breathing bugs (water-boatman, backswimmer, water strider) may become abundant.
6	Fish populations are reduced or eliminated.

Saprophytes are plants, like mushrooms, which derive their nourishment from decaying organic matter. Ciliate fauna is similar in analogy to a class of protozoan having numerous cilia (e.g. eyelash protozoan).

As the USEPA study indicated, typically pH values found at AMD sites are alone capable of decreasing biotic populations to levels observed in Leading Creek, regardless of the added presence of AMD toxicants, fine flocs, turbidity, and heavy sand sedimentation attributed to AML. This finding was supported by the initial field surveys that showed a predominant sensitivity of instream biota to AMD acidification (e.g. lowered pH trends with low biodiversity). It is critical to realize that pH can remain an overall good predictor of adverse impacts to biota in large-scale studies (Allan 1995; Gallagher *et al*, 1999).

Table 2.32 - Summary of Damages to Aquatic Organisms With Decreasing pH;
(Adapted from USEPA, 1980)

pH	Biological Effects
8.0 – 6.0	Long-term changes of less than 0.5 units are likely to alter the biotic composition of freshwaters to some degree. The significance of these slight changes, however, is not great. A decrease of 0.5 to 1.0 pH units in the range of 8.0 to 6.0 may cause detectable alterations in community composition. Productivity of competing organisms will vary. Some species will be eliminated.
5.5 – 5.0	Many species will be eliminated, and species numbers and diversity indices will be reduced. Crustacean zooplankton, phytoplankton, molluscs, amphipods, most mayfly species, and some stone fly species will begin to dropout. In contrast, several invertebrates tolerant of pH will become abundant, especially the air breathing forms (e.g. *Gyrinidae*, *Notonectidae*, *Corixidae*), those with tough cuticles which prevent ion losses (i.e. *Sialis lutaria*), and some forms which live within the sediments (*Oligochaeta*, *Chironomidae*, and *Tubificidae*). Overall, invertebrate biomass will be greatly reduced.
5.0 – 4.5	Decomposition of organic detritus will be severely impaired. Autochthonous and allochthonous debris will accumulate rapidly. Most fish species will be eliminated.
< 4.5	All of the above changes will be greatly exacerbated, and all fish will be eliminated.

The following notes compiled in Table 2.33 on associated impacts to fish populations as a result of instream acidification were also presented by USEPA. In that study, a generic process of instream acidification was broken down into three stages and can be viewed as a continuum of water quality change paralleling an acidimetric titration of available bicarbonate (Snoeyink *et al*, 1980; USEPA, 1980).

Table 2.33 - Regional Assessment of Acidification Impacts of Fish Populations
(Adapted from USEPA, 1980)

Acidification Stage	Instream Effects
1	Decreased alkalinity occurs but pH remains above 6.0, and bicarbonate buffering is maintained. No significant impacts to fish population are observed at this stage.
2	Loss of HCO_3 buffering occurs, resulting in severe temporal fluctuations in pH. During this stage, stress, reproductive inhibition, and episodic mortality may initiate recruitment failure and eventual extinction of fish populations.
3	Final stage of acidification is characterized by chronically depressed pH and elevated heavy metals. Fish are generally absent from waters at this stage.

2.12 LEADING CREEK SITE RECONNAISSANCE

Preliminary site investigation indicated that the Leading Creek stream system studied here (38,800 ha, 96,000 acres, 388km^2 or 150 square miles) experiences significantly varied flow regimes under low, medium, and high discharge conditions. The hydraulic influence of the Ohio River on the stream system under flood and non-flood conditions is also evident. This is particularly true at the lower reaches of Leading Creek and Thomas Fork, and up through the Corn

Hollow area towards Dexter, Ohio during extreme flood conditions (up to and above RKM 17 to 25 depending on severity).

Initial concerns associated with hydrologic and sediment/contaminant assessment of the stream system were guided by the following preliminary observations (Cherry *et al*, 1999):

- Leading Creek appears to exhibit a diverse range of flow, morphological, and depositional characteristics expected in a natural alluvial system, where a variety of hydrologic and subbasin-scale influences are apparent within different stream reaches.

- Ecosystem responses to stress appear to be due in part or in whole to either soluble chemicals or toxic sediment. Poor biotic responses also appear due in part to direct physical impairment or displacement of biota caused by instream or riparian habitat degradation and other changes in land use/cover.

- The upper half of the stream system (above Dexter, Ohio) appears to be potentially influenced by sediment/contaminant loading from agricultural inputs, and to a lesser extent, older abandoned strip mine lands (ASML).

- The relative impact of loading influences is likely to reverse roles towards the lower half of the stream system where older and younger ASML become more prevalent. In lower tributaries and reaches ASML along with historic abandoned underground coal mining (AUML) dominate acid/base equilibrium conditions. Severe AMD occurs with pH <4.

- Though harder to discern with commingled AML, nearby, agricultural and other miscellaneous land use/cover influences likely play a key role in ecosystem health along tributaries and mainstem areas in the upper and lower parts of the basin.

- Direct physical impacts upon the ecosystem from sedimentation are severe in many tributaries and along most of the lower half of the Leading Creek's mainstem. Sediment loading impacts appear to be derived from siltation from AG and sand sedimentation from upstream ASML.

- Upstream headwater tributary areas are equally under strong influence by urbanization and agricultural practices that lead to excessive siltation and poor physical and chemical water quality in some tributaries.

- Critical mainstem habitat appears impacted by sediment deposition associated with the presence of both AML, AG, and urban activities. Watershed-wide sedimentation has been greatly exasperated by overall changes in land use/cover over the last 50 years.

- AML sedimentation is at least a primary cause of continued impairment to ecosystem health and exacerbates flooding. Attributed to AML, Leading Creek has been ranked as one of the most heavily inundated sediment-laden streams in all of Ohio.

- ♦ Generally accepted practices for sediment and erosion control in the watershed appear to be poor or non-existent in many cases, including but not limited to AML, AG, urbanization, smaller "independent" timber operations, and road maintenance activities.

- ♦ The interaction of corridor activities, channel activities, and residual bed contaminants transported down the stream system, join with continued non-point and point source discharges urban AG, AML entities are seen as the greatest elements of preliminary concern in the risk assessment. These may significantly influence physical, chemical, and biological conditions throughout the Leading Creek Watershed.

Where agricultural influences are mentioned above, the potential influences of other land use activities including forestry, industry/commerce, and residential land use/development were considered potentially significant in certain areas. Finally, along with AML effects noted in the watershed from surface or strip mining (ASML), residual materials left at the surface of abandoned underground mines (AUML) might also contribute significant sediment/contaminant loading in some areas. Prior to GIS work conducted here, differences between physical and chemical impacts upon biota form ASML and AUML was previously not well characterized.

2.13 RISKS TO AQUATIC LIFE IN LEADING CREEK

Primary features of the two OEPA reports (OEPA, 1991a, 1991b) representing background ecological attributes of Leading Creek are summarized below, incorporating other salient mechanisms, notes, and preliminary risk assessments. The resulting integration of the previous assembled "baseline" studies provides an outline and prioritization of what potentially may be most important aspects in assessing risks through hydroecological assessment of Leading Creek. The two OEPA studies and other background review indicated, in general, the following anthropogenic influences with the potential to adversely impact water quality and biota (Cherry *et al*, 1999; OEPA, 1991a, 1991b; USDA, 1983, 1985). Listed in an approximate order of initially presumed importance, some statements are scale dependent and require appropriately qualified interpretation.

Summary of Potential Risks to Aquatic Life in Leading Creek

- ➢ **Coal mining operations in the Western Allegheny Plateau Ecoregion:**

  - ▪ Erosion and sedimentation from spoil piles and open pits,
  - ▪ Introduction of toxic compounds,
  - ▪ Impacted water quality, and
  - ▪ Direct physical disruption of the streambed.

- ➢ **AML Sediment Cycle, Exasperated by Floodplain Encroachment:**

  - ▪ Accelerated upland erosion and instream sedimentation of fines/sands,
  - ▪ Excessive bed loads, aggradation, degradation, and channel slope fluctuations,
  - ▪ Seriously destabilized banks, particularly in areas already under increased stress from denuded riparian zones in the floodplain, and

- Bank instability within the unglaciated channel-valley floodplains tends to occur more often at lower elevations of watershed relief where AML-sands also prevail. In addition stability is degraded on specific reaches associated with active mines and activities by a handful of farmers currently attempting channelization.

➢ **AML/AMD Impact Patterns:**

- Simple acidification of streams can impact biota and degrade community structure to levels typically observed in AML/AMD impacted areas,
- AMD is accompanied by chemical flocs (e.g. yellow-boy and snowflake, etc.), turbidity , and high concentrations of soluble heavy metals, iron and aluminum,
- OEPA noted extremely high iron concentrations at sites severely impacted by AMD run-off; aluminum, manganese, nickel, zinc, copper, and arsenic are also common pollutants,
- Adverse AMD impact in Leading Creek was primarily limited to tributaries joining the lower mainstem where it is dominated by sand beds, and
- The most toxic run-off conditions appear to occur with spring flows, though toxicity can prevail throughout the year under all flow conditions.

➢ **Crops and Livestock Production:**

- Amplified stresses due to locally dominant narrow valleys,
- Removal of stream-bank vegetation to extend cropland and pastures,
- Increases in channelization and bank modification,
- Encroachment upon riparian zones, sometimes plowing directly into channels, and
- Incurred stresses cause stream bank erosion, turbidity, and sedimentation.

➢ **Agricultural/Silviculture Siltation Impact Patterns:**

- Turbidity and substrate embeddedness,
- Silviculture also expresses similar instream sedimentation characteristics,
- Large numbers of exceedences of iron water quality criteria are common for Ohio streams (OEPA, 1991a, 1991b) and were attributed by OEPA to the high iron content of local geology, this would be presumed to occur as non-soluble suspended washload due to high levels of iron found,
- According to OEPA, ubiquitous background concentrations of elevated iron associated with non-AMD land uses rarely result in observable impairment of the instream biota (OEPA, 1991a, 1991b), and
- Coliforms and toxicity can accompany agricultural crop and livestock operations, though siltation is by far the most frequent concern noted.

➢ **Active Underground Long Wall Mining and Associated AMD Impacts:**

- Potential residual impacts of emergency dewatering of SOCCO Meigs Mine No. 31 in the summer of 1993,

- Effluents from wastewater treatment operations at coal preparation plants serve as groundwater diversions into receiving basins, and
- Effluent discharges contain modest concentrations of Fe and Mn, low heavy metals, elevated levels of soluble salts (chlorides, sodium, and sulfates), high alkalinity, and correspondingly elevated levels of specific conductivity (USEPA, 1976a).

➢ **Urban Run-Off, Localized STP Impacts, and Historical Channellization**

- Lower dissolved oxygen levels and nutrient loading downstream of STPs,
- Impacts of urban run-off and commonly associated severe channelization, and
- Sources of coliforms identified included municipal waste treatment systems, urban run-off, and on-site disposal systems (septic fields).

➢ **Gas and Oil Well Production Operations**

- Oil and brine contaminate surface and groundwater,
- Improperly plugged wells, overflow and spills, and
- Potentially impacted instream sediment and water quality.

Historical problems associated with resource extraction in Leading Creek have been noted. Previous biological studies have been conducted inside the watershed, and in similar basins nearby. Along with concerns occasionally noted for agriculture, urban, and timber operations, these studies consistently identified AML risks and associated instream impairment as the ecoregion's greatest obvious and continuing concern (OEPA, 1991a, 1991b; USDA, 1985). Next to severe AML-AMD problems Ohio's streams are plagued by urbanization and agricultural operations where sedimentation plays a significant, deleterious role in decline of stream health (Yoder *et al*, 1996).

2.14 RIVER MECHANICS AND SEDIMENT TRANSPORT

River mechanics, water flow, sedimentation and channel stability are topics of great importance in understanding man's influence on aquatic ecology (Allan, 1995; Waters, 1995).

Most streambed "forms" or bed-forms typically associated with alluvial streams are present somewhere in Leading Creek under various flow regimes. Bed-forms include ripples, dunes, anti-dunes, bars, flat beds, glides, chutes and pools (Yang, 1996; Vanoni, 1977). Bed forms represent a primary structural element of resistance in the channel. This structure is likely obliterated at conditions nearing bankfull and during flash flooding. The bed-forms eventually re-establish as flooding recedes (Diplas *et al*, 1992b, 1994, 1997; Milhous, 1982, 1998).

In gravel bed and sand bed dominated streams, it has been estimated that ~99% of sediment by weight is carried during storm events that comprise less than ~3% of the time record (Diplas, 1997; Leopold *et al*, 1964). This is to say that most sediment moves during bankfull, or near bankfull conditions. Some researchers estimate the threshold to range down to about 80% of bankfull condition (Diplas, 1997; ; Lisle, 1989; Parker *et al*, 1982). At some level anyway a significant jump in carrying capacity is presumably realized by natures design, as the bed is mobilized, or fluidized. Streambed forms, as resistance to flow, dissipate into a more homogeneous conveyance structure at this near bankfull condition (e.g. analogously, a smoother pipe is temporarily formed which allows even more materials to pass) (Diplas, 1994, 1997).

2.14.1 Overview of Stream System Stability

Sediment-water interaction observed in Leading Creek presents an anticipated stream dynamic of two-phase flow. The two phases, liquid-water and solid-sediment, are in constant dynamic harmony (Diplas, 1997). Bed load, suspended load, and washload involve transport, bed and bank scour, and bed deposition mechanisms (Diplas, 1987; Klingeman *et al*, 1982, 1993; Parker *et al*, 1982; Vanoni, 1977; Yang, 1996). These mechanisms are continually optimized to deal with variable water and sediment inputs from surface run-off and channel storage. Perturbation analysis can provide a useful point of view in understanding stability issues, as can unit stream power approaches, both imparting that a great deal of energy is in balance (Diplas, 1997; Yang, 1996).

On human time scales, geomorphologic balance is not a simple task to quickly realign, once disturbed (Leopold *et al*, 1964; Rosgen, 1996). If the channel-valley unit remains intact, proper hillside unit management can provide remarkably rapid responses to restoration efforts of *in situ* problems (e.g. like too much sand or silt or too much of both) (USEPA, 1998). Hillside management likely has a comparably reduced cost compared to structural control of the stream's plan form geometry, for similar benefit. In-channel control structures may add to total bank resistance reducing channel sinuosity, initially causing reduced sediment flow, but ultimately leading to steeper slopes. The process can result in further upstream aggradation, and downstream degradation (Lane, 1955; Leopold *et al*, 1964; Rosgen, 1996). Proper landscape management as opposed to channel manipulation is a more viable long-term, natural approach to achieving stability.

Attempting to assess the stability of 750 km of streambed in the Leading Creek Watershed, each reach with a different set of circumstances, presents inherent limitations, and impracticality. One can feasibly generalize to applications of stream mechanics concepts like Lane's equation and to other bankfull discharge concepts like effective or dominant discharge (Goodwin, 1998; Hey *et al*, 1998; Lane, 1955; Rosgen, 1996). These might logically be applied first at key cross-section locations on the mainstem of any system where severe problems are noted. Like watershed risk assessment, attempts to fix, modify, or otherwise control a single reach requires holistic analysis, addressing all upstream concerns (NRC, 1992; Rosgen, 1996; USDA, 1998).

2.14.2 Gravel Bed Streams and Sediment Transport

Transport of sediment is based on concepts of turbulence and shear exerted in channels with erodible boundaries (Bedient *et al*, 1992; Linsley *et al*, 1975; Vanoni, 1977; Yang, 1996). For example, deposition or scour of cohesive materials can be based on parameter estimates that account for the electrochemical nature of clay minerals and available shear stress (Mitchell, 1993; Sharma, 1994). Example formulations that address shear stress of colloidal sediments are the Krone and Partheniades equations used in HSPF (Bicknell *et al*, 1993). For gravel and sand bed streams, the bed still represents an erodible boundary. The conditions for incipient motion are guided by physical and mechanical processes only, with the application of equations of continuity and some assumption based on momentum, for instance uniform flow used by Manning (Vanoni, 1977; Yang, 1996).

One thing that may be quickly appreciated in stream mechanics is a certain lack of theoretical basis associated with many equations used to predict flow of water and sediment (Diplas, 1997). Most equations are empirical, though similarities obviously exist among different equations. Particularly lacking for gravel bed systems is accurately measured bedload data (Ashida *et al*, 1972; Diplas *et al*, 1987, 1997; Garcia *et al*, 1991; Gomez *et al*, 1989; Meyer-Peter *et al*, 1948; Parker *et al*, 1982; Yang *et al*, 1996).

2.14.3 Bed Resistance and Particle Size

For gravel beds, Manning's equation is satisfactory for estimating bed resistance and associated water depth for a given flow and cross-section (Vanoni, 1977; Yang, 1996). Bed resistance in gravel beds (e.g. Manning's n value in Manning's uniform flow equation) is often calculated as a function of the pavement D_{90} (Diplas *et al*, 1987, 1992b, 1994; Einstein, 1950; Klingeman *et al*, 1982; Parker *et al*, 1982; Lisle, 1989).

For gravel beds, resistance will be primarily determined through pavement particle size. For sands, both frictional resistance of water flow and resistance to bed forms are typically needed to assess the stage flow relationship, if flow is not directly measured (Vanoni, 1977; Yang, 1996). For sand and gravel beds, a jump in transmission capacity of the channel occurs as the bed materials reach mobility or "near-mobility", a threshold concept defining sediment transport conditions (e.g. the incipient motion criteria for grains). The obliteration of internal channel structure and bed forms, to whatever degree it occurs under conditions well above threshold shear stress, has an overall effect that tends to temporarily flatten bed forms (Diplas, 1994). This allows nature to maintain smaller channels for the same loads of water and sediment (Diplas, 1997; Waters, 1995). Stability and stream mechanics represent a critical physical interaction in which biological systems have evolved to accommodate these functions.

2.14.4 Incipient Motion Criteria

The incipient motion criteria or mobility threshold is exceeded at some point under ever increasing shear stresses acting on the bed (Garcia *et al*, 1991; Diplas *et al*, 1987, 1992b, 1994; Parker *et al*, 1982; Klingeman *et al*, 1982, 1993). Under uniform flow conditions, which prevail for the most part across Leading Creek, the increased shear stresses are due to surface run-off and the subsequent increased water depth in the channel (also referred to as increased head or stage). This is only to say that shear stress under uniform flow implies constant energy slope. Actual shear stress under uniform flow is determined by depth, being defined by depth*slope/(R*D_{50}). R is the submerged weight for the sediment in water, 1.65 for quartz and D_{50} is the mean particle diameter size (Yang, 1996).

In concept, at some point shear stress exceeds a local critical shear stress, and a particle is mobilized, a concept of "excess shear". As estimated by Shields, gravel beds are not normally considered Reynolds Number dependent, but sand beds are Reynolds Number dependent. For gravel beds, critical dimensionless shear stress is typically ~0.03 to 0.06. For finer particles, mean particle size determines the boundary Reynolds number from which a dimensionless critical excess shear stress can be estimated via Shield's curve (Vanoni, 1977; Yang, 1996).

2.14.5 Gravel Bed Framework

For gravel beds, the tougher pavement skin or "armor" of a riffle bar is larger in particle size than the sub-pavement material below it, and must be mobilized first (Andrews *et al*, 1987; Parker *et al*, 1982). This will occur though at a much higher shear stress than would be normally realized for the overall average unsorted parent bed material. Gravel bed transport is often broken down by fractions of particle size, where each fraction and associated incipient motion criteria are analyzed individually, and then summed (Ashworth *et al*, 1989; Diplas *et al*, 1987, 1992b; Einstein, 1950). This natural sorting mechanism and creation of pavement and subpavement (Andrews *et al*, 1987; Lisle, 1989) is directly evident in most areas of a riffle bar (Diplas *et al*, 1992b, 1994). At full bankfull stage some researchers also conceive of vertically separated sheets of gravel flow based on particle size (Milhous, 1982). This concept, a more discrete model, is somewhat analogous to that developed for suspended sands and silts (suspended load equation) by Rouse. The concept is intuitive, and implies that under conditions of sufficient shear stress, the gravel bed liquefies, stratifying into vertically separated horizons of different sized gravel particles, moving downstream (Yang, 1996).

For all attempts to describe incipient motion criteria for gravel beds, the concept of near-equal mobility would appear to be the most accurate, physically based model available today. Within the pavement of gravel beds, the concept of near-equal mobility implies that the point at which a given particle size range actually mobilizes as bed load is dependent on the size of materials around it (Diplas, 1987; Diplas *et al*, 1992b; Parker *et al*, 1982). Smaller sized sub-pavement particles below a completely mobilized pavement zone are considered to entrain simultaneously. An armored pavement layer, a subpavement layer, and a bottom layer define structure of the gravel-bed framework. The pavement and subpavement bottom layers are roughly deemed D_{90} thick, and 2* D90 thick, respectively, with an underlying bottom layer representing overall parent material (Diplas *et al*, 1992b, 1994; Lisle 1989; Adams *et al*, 1980). Typical relations among D_{50} for the subpavement layer, bottom layer, and overall transported bedload are given by $D_{50\text{-subpavement}}. \cong D_{50\text{-bottom}} \cong 1.14 D_{50\text{-bedload}}$ (Diplas *et al*, 1992b).

2.14.5.1 Subpavement Characteristics

After flooding recedes and the bed framework is re-established with particles >0.125mm, subsequent increases in materials < 0.125mm can occur until the framework is again mobilized, or until the bed becomes completely inundated by fines. This increase in fine material comprising the bed matrix will only occur above a depth referred to as the seal depth. The subpavement skin or seal thickness is perhaps ~ 2.5-3*D_{90} (up to 5*D_{90}) of the framework material and is created by fines (<0.125 mm) that have worked their way through the viscous layer down into the gravel bar itself (Lisle, 1989; Diplas, 1994). Aside from fines loading, the make-up of gravel framework particle size distribution otherwise does not change remarkably (Diplas *et al*, 1992b, 1994; Lisle, 1989; Parker *et al*, 1982). Just below critical shear stresses for the pavement, bed materials >0.125mm behave as bed load and suspended load. These larger size particles would bridge before infiltrating further down into the subpavement zones in any greater concentration. The seal effect describes the point at which bridging occurs for fine materials like silt and clay (Diplas, 1994; Lisle, 1989).

Summarizing, the layered structure of the inter-gravel framework (materials >0.125 mm) is determined by (a) sediment size distribution at the time the hydrograph recedes below critical flow conditions, and (b) fine material loading in the interim until the pavement is once again mobilized by shear stress. For these reasons, accurate particle size distribution sampling of gravel beds is key to understanding most stream mechanics problems associated with them (Diplas *et al*, 1992a; 1997). As Diplas has shown, accurate volumetrically equivalent measurement of particle size distributions of gravel bed layers is non-trivial and can be far more complex than sampling sand beds or conducting pebble counts (Wolman, 1954).

2.14.6 Sand Bed Streams and Sediment Transport

Sand bed systems do not experience armoring and associated pavement conditions (Diplas, 1997; Yang, 1996). There is little if any significant particle size variation in the vertical bed profile. Lateral heterogeneity can remain profound and longitudinal homogeneity often prevails, at least within a given reach (Diplas, 1997). Common bed form structures are ripples (only for materials <0.6 to 0.9mm), dunes, and bar storage (Vanoni, 1977; Yang, 1996). In sand beds, two forms of resistance, particle resistance and bed form resistance, control water level. At a known cross-section, each form of resistance can be estimated for its contribution to stream depth for a given flow (Vanoni, 1977).

At transitional stages above the incipient motion criteria for individual sand particles, bed form structure and associated resistance becomes more homogenous (Vanoni, 1977; Yang, 1996). Below this threshold, sand is in transport, but bed form resistance is still high. Above this threshold, bed forms eventually are completely obliterated, and a flat bed condition will prevail towards increasingly greater flows or “upper regime” flow conditions. The effect is reduced net bed form resistance longitudinally which maximizes channel carrying capacity (Diplas *et al*, 1992b, 1994).

Sand bed sediment transport or suspended load transport can be based on Rouse’s work which allowed separation of bed load from suspended load. Later Brooks successfully simplified this work for computational purposes (e.g. Brooks Method). In summary, using the Rouse equation, a vertical gradation in “suspended” materials is realized, based on particle size. The concentration profile in the depth vertical is not constant (Vanoni, 1977; Yang, 1996). At upper regime stream velocities, eventually the stream bed-form progression develops sinusoidal bed (and surface) wave forms or anti-dunes, and at still higher flows the bed form progression will develop to a configuration of chutes and pools. This latter bed form progression at higher flows may be applicable to both sand bed and gravel bed streams (Vanoni, 1977; Yang, 1996).

2.14.7 Washload, Suspended-Load, Bed-Load, and Dissolved-Load

2.14.7.1 Washload (Silt and Clay)

Clay, silt and very fine sand loads are transported as total washload. Washload can usually be expressed as a relatively constant concentration of suspended solids across vertical and lateral profiles for any stream cross-section point (Vanoni, 1977; Yang, 1996). Non-cohesive medium and coarse silt and very fine sands (>0.008mm, <0.125mm in particle diameter) and cohesive fine silts and clays (<0.008mm) define the washload range of particle diameter sizes. Materials smaller than 0.00045 mm or 0.45 microns are by definition described as soluble materials. The cohesive characteristic can be defined in the range of 2 to 10 microns.

Washload is transported downstream with water currents until stream conditions allow for partial settling and re-sedimentation, such as pools and mouth/delta regions. Only low velocities and minor turbulence are required to keep washload in suspension (Diplas *et al*, 1992b; 1994). Under sub-critical flow conditions in gravel bed channels, fines can also fill inter-gravel pore spaces of riffle bars and pools (Diplas *et al*, 1992b; Lisle, 1989). Washload is an important concept in overall bank stabilization for its “sticky” quality. Cohesive materials allow for vertical and undercut banks, an otherwise unstable configuration for non-cohesive sediments.

Transport of washload occurs independent of shear stresses normally used to determine effective transport of gravel (transported as bed-load) and sand (transported as suspended load) (Bicknell *et al*, 1993; Vanoni, 1977; Yang, 1996). Washload is the finer material that always remains in physical suspension above the channel bed once it is scoured. Washload constitutes typically a very small fraction of overall bed materials (Diplas *et al*, 1992b).

A critical point of its associated mechanics, washload is for all practical purposes supply limited, not stream capacity limited. The stream has an almost infinite capacity to carry ever-increasing loads of silt and clay supplied from the hillside and channel-valley units (Bicknell *et al*, 1993; Leopold *et al*, 1964; Vanoni, 1977; Yang, 1996). A gravel bed or sand bed stream can be limited by sediment supply (bed, bank, and upstream sources) or its sediment carrying capacity (Bicknell *et al*, 1993; Rosgen, 1996; Vanoni, 1977; Yang, 1996).

High washload can typify stream systems with a high % silt-clay content in the bank material, as in Leading Creek (Diplas; 1997). Also typified by high washload are subsheds with significant widespread agricultural practices or other disturbances that alter soils with high % silt-clay content, including road and dam building (Curtis, 1973, 1979; Dickens *et al*, 1985, 1989; USDA, 1998). Washload can induce significant impacts upon larger bed material (like riffle bars) under extreme loading conditions, or in areas dominated by fine sediments such as reservoirs and lakes (Bedient *et al*, 1992; Diplas *et al,* 1992b, 1994; Linsley *et al,* 1975).

Depending on loading rates and duration, loading of fines into gravel streambeds and associated inter-gravel matrices can be significant under conditions below effective pavement (i.e. armorcoat) mobility (Adams *et al*, 1980; Bradley *et al*, 1991; Diplas *et al*, 1987, 1992a, 1992b, 1994; Fripp *et al*, 1993; Garcia *et al*, 1991; Lisle, 1989; Parker *et al*, 1982; Shirazi *et al*, 1981). Leading hysteresis effects on stream hydrographs will also likely be observed with washload monitoring due to higher initial supplies of silt/clay, both instream and on land (Bedient *et al*, 1992; Cluer, 1998; Linsley *et al,* 1975).

2.14.7.2 Suspended-Load (Sand)

For typical Total Suspended Solids (TSS) monitoring, unless collected with a multi-level, flow proportional sampler, results are associated with silt and clay fractions representing washload (e.g. via Rouse equation). TSS monitoring does not actually physically monitor the normally defined range of suspended-load sediment, which runs from approximately 0.125mm to 2mm (fine to coarse sands). Suspended load is material held in suspension by turbulence, travels at about the same rate of local average water velocity, and can comprise a significant amount of overall bed material (Diplas, 1997; Vanoni, 1977; Yang, 1996). Suspended load transport for this range in most cases can be related to a power function of discharge (Yang, 1996). Equations found in literature may or may not incorporate washload concepts with suspended-load transport concepts (Yang, 1996).

2.14.7.3 Bed-Load (Gravel)

Covering ranges of particles that slide, roll, or saltate at or very close to the channel bed (i.e. particles that hop, skip, slide, and roll), bed load is typically considered to be comprised of boulder, cobble, and gravel particle sizes, and also includes coarser sands. Bed-load transport for this range is in simple conceptual cases partially equivalent to a power function of discharge (Bicknell *et al*, 1993; Lisle, 1989; Yang, 1996). An accurate assessment of bedload can be very difficult to arrive at both theoretically and in direct measurement (Diplas, 1987; Einstein, 1950; Glysson, 1993; Gomez *et al*, 1989; Garcia *et al*, 1991; Klingeman *et al*, 1982; Parker *et al*, 1982)

2.14.7.4 Dissolved Load

Materials smaller than 0.45 microns entrained in the water column are by definition described as soluble or dissolved materials. To distinguish between suspended load and soluble load in a water sample, the sample must be physically filtered. If looking at dissolved metals, this must obviously occur prior to preservation of the samples. Regarding typical total load of a stream, the soluble load on average across the world is 38% of the total load. This would also be close to specific values predicted based on climatological indicators (Diplas, 1997; Dunne *et al*, 1978; Leopold *et al*, 1964).

2.14.7.5 Sediment Load Summary

An important concept for gravel, sand, silt, and clay, even though they are composed of similar materials (Sharma, 1994), they have fundamentally different stream mechanics (Bicknell *et al*, 1993; Vanoni, 1977; Yang, 1996). Aspects of stream mechanics defining bedload, suspended load, and washload in many ways drive the separation of particle diameter sizes discussed. Transport behavior for very fine silt and clay is less associated with gravitational sedimentation, and more dependent upon the electrochemical properties associated with colloidal behavior (Mitchell, 1993; Sharma, 1994).

2.14.8 Sediment Transport

For coarser, non-cohesive materials characteristic of bed load and suspended load transport (e.g. sediment > 0.1 mm), H.A. Einstein, son of Albert Einstein, may be considered the greatest single researcher in stream mechanics (Diplas, 1997). Einstein developed a combined bed load and suspended load transport function, based partially on theoretical evidence. His fractional bed load transport equation advanced the concept that size ranges of particles could be looked at on a fractional basis (Einstein, 1950). Fundamental to his approach, Einstein abandoned excess shear concepts. Excess shear assumes that nothing on the bed moves until everything moves. From this ensued the near-equal mobility concepts advocated more today. All of these equations basically entail gaining knowledge of slope, water depth, and particle size from which shear stress is determined.

Einstein showed that transport equations could be integrated by summing individual results by their relative % by weight found in the original bed material, e.g. summed from individual calculations completed for each size range. This aspect of superposition allowed address of many subtle mechanisms more directly related to each particle size class mentioned. Einstein himself invented a hiding function to address aspects of unequal mobility between various size fractions. Other researchers have extended sediment transport research more recently in an attempt to continue to unify available sediment transport equations for gravel bedded streams (Diplas *et al*, 1987, 1992b; Garcia *et al*, 1991; Parker *et al*, 1982). Most bed load equations are based solely on empirical evidence, for which only sparse, acceptable calibration data exists due to the extreme difficulty in accurately measuring bed load (Diplas, 1997; Glysson, 1993; Milhous, 1982).

2.14.8.1 Sediment Transport Equations

For sands, procedures are well established based on combinations and variations of using Einstein's method together with Rouse's equation, for example employing either Toffaleti's method or Colby's method (Bicknell *et al*, 1993, Donigian *et al*, 1995; Yang, 1996). Rouse's equation describes vertical concentration gradients for suspended materials based on particle size (Vanoni, 1977; Yang, 1996). There are ongoing arguments as to which are the best empirical transport equations for which situation (Ashida *et al*, 1972; Gomez *et al*, 1989; Garcia *et al*, 1991; Meyer-Peter *et al*, 1948; Yang, 1996). Of interest is to what degree do transport equations address the above-described models, and to what degree do these models appropriately describe real, underlying mechanisms and stream mechanics for both sand and gravel bed streams. For the same stream conditions, bedload estimates can range four orders of magnitude among the existing sediment transport equations offered in the literature (Parker *et al*, 1982).

The conceptual model of gravel bed transport based primarily on Einstein's bedload function extended with concepts of equal mobility (Parker *et al*, 1982; Andrews *et al*, 1987), and, more lately near equal mobility, has both a strong theoretical basis and some experimental confirmation (Einstein, 1950; Garcia *et al*, 1991; Diplas *et al*, 1987, 1992b, 1994). Of other historical conventional bedload equations available, the more accurate and most applicable to stream conditions in Leading Creek would be the Ashida/Michiue equation and the Peter-Meyer/Muller transport equations (Ashida *et al*, 1972; Meyer-Peter *et al*, 1948; Yang 1996).

2.14.9 Bankfull Design Conditions

As a simplification of the world, a single design parameter is often desirable to describe complicated phenomena. For example, a 7Q10 flow is used to evaluate mixing of a contaminant instream and to understand its effect on biota at critical low flow conditions. The 7Q10 is the lowest 7-day average flow annually that reoccurs once every ten years, on a statistical basis. An analogy for flooding, channel maintenance, and understanding critical stream mechanics principles is the natural bankfull condition. An important concept recognized in the latter decades of the century is the natural undisturbed bankfull design condition analogous to the effective discharge condition, channel forming condition, or dominant discharge condition (Diplas, 1997; Dunne *et al*, 1978; Goodwin, 1998; Hey *et al*, 1998; Leopold *et al*, 1964; Rosgen, 1996). It is well established that size, shape, and pattern of rivers and channels are related to the flows carried (Fetter, 1994; Leopold *et al*, 1964; Rosgen, 1996; Strahler, 1957).

It is believed that the dominant discharge condition can succinctly represent a great deal of the systems overall composite morphogenetic trait. As discussed in Leading Creek's preliminary assessment for the Ohio area, a typical long-term bankfull design recurrence interval of 1.5 to 2 years would be expected to prevail (Dunne *et al*, 1978; Leopold *et al*, 1964; Rosgen, 1996). The bankfull design flow is a similar concept in stream mechanics to a 7Q10 for example in that it represents, in this case, a useful gage of the dominant mechanisms that appear to form channels, accounting for local climate via a duration term. Bankfull condition is sometimes used as a check of stability, though it doesn't replace other standard channel design and assessment methods (Hey *et al*, 1998). Bankfull design is best seen as a single tool in a tool-bag of tools that can evaluate stability issues associated with the plan form geometry of rivers and streams.

Advocated now by the Army Corp of Engineers, regional effective discharge curves can be developed and used as a parallel geomorphic design tool. These are evaluated by determining a site's associated bankfull width-depth ratio and placing it against regional "reference" measures of the same (Rosgen, 1996). If bankfull width-depth deviates significantly from that of an undisturbed reference stream of the same type, class and particle size, severe structural problems may be indicated. Humans most often notice channel instabilities or departures from nature's dynamically stable plan-form design during or as a result of major flood events.

2.14.9.1 Pitfalls of Using Extreme Events for Channel Design

Throughout the world, a common pitfall in stream engineering this century appears to have occurred in designing channels for flood flows. The drawbacks realized are associated with the use of longer duration flood reoccurrence intervals of 10, 25 or 50 years, as opposed to bankfull reoccurrence every 1.5 to 2 years. Typically return period flows are based on a Pearson-Log distribution assumption (Bedient *et al*, 1992; Linsley *et al*,1975).

Problems arose because sediment transport is primarily determined by the more frequent but less dramatic bankfull and near-bankfull conditions (Leopold *et al*, 1964). Severe problems including lateral migration, entrenchment, and increased flooding have been associated with excessive channelization and flood plain encroachment (Rosgen, 1996). This occurs when nature has been unduly restricted in its ability to store and transport sediment and water, or is otherwise restricted in plan form to accommodate changes in loading from the hillside areas (Dunne *et al*, 1978; Leopold *et al*, 1964; Rosgen, 1996; USEPA, 1998; USDA, 1998).

The bankfull design concept places useful boundaries on what one can expect a given morphology to carry in terms of water and sediment loads (Rosgen, 1996), and should be checked for convergence with more formal engineering design approaches (Diplas, 1997). If balanced parameters are disrupted, instability may proceed with potentially increased flooding, increased bank erosion, and increased meandering. It many cases a stream may be perceived to be unstable (e.g. flooding, bank erosion, and meandering), when in effect the streams response is moving the channel towards a desired stabilization. One way or the other, humans living in the floodplain or attempting to determine plan form geometry of channels will tend to remain at great odds with the river's energy.

2.14.9.2 Channel Design Summary

In summary, of most important note is that bankfull and near bankfull magnitude and frequency determine geomorphologic structure, sediment transport, and depositional/scour patterns. It is not events of catastrophic proportion, but those of moderate proportion and frequency that account for much of the sediment transported by a stream (Leopold *et al*, 1964; Rosgen, 1996; Diplas *et al*, 1992b). Engineering of natural channels is most efficiently arranged by nature where only moderate modifications should be attempted, if any. Analysis of stability of plan form geometry and re-engineering of channels should be investigated on a reach by reach basis (Diplas, 1997; Rosgen, 1996).

2.14.10 Stream Aggradation and Degradation

2.14.10.1 Predicting Hydraulic Changes

Plan form instability increases under stresses due to channelization, increased water flow, and increased sediment flow (Diplas, 1997; Vanoni, 1977; Yang, 1996). Lane's equation (Lane, 1955; Rosgen, 1996; Waters, 1995), described below, is useful in predicting general trends in stream response to short-term and long-term changes in load of water or sediment to the stream channel. Lane's equation is given by:

$$Q_{sediment} * D_{50} \sim \text{is proportional to} \sim Q_{flow} * S_{energy}$$

Where:

- sediment load ($Q_{sediment}$)
- water flow(Q_{flow}),
- energy slope (S_{energy}), and
- particle size (D_{50}).

The above equation is useful for evaluating generic trends. It is helpful to realize that most often sediment load and water-flow represent long-term independent variables. For the short-term, slope and water flow are likely the independent variables (Diplas, 1997). In most situations, sediment load and water-flow tend to move together, not in isolation (Diplas, 1997).

Typically, a great deal of historical data must be collected to accurately determine if a given stream reach is aggrading or degrading. It can be a difficult task, where the primary questions lie in determining the landscape's limiting factor or controlling mechanism (Diplas, 1997; Lane, 1955; Rosgen, 1996; Galay, 1983).

2.14.10.2 Aggradation

The end long-term results of increased sediment discharges from the land surface will usually be aggradation, a possible increase in slope, and a significant decrease in particle size (Diplas, 1997; Lane, 1955; Galay, 1983). This will lead to reduction of the heterogeneity and dynamic velocity configurations otherwise available.

Aggradation, or the accumulation of valley alluvium, will raise the channel bed, and can be a geomorphic response to many changes including tectonic and climatic changes (Leopold *et al*, 1964). It can also be a more recent response to changes in flow of water and sediment induced by man, for example upstream of a dam (Diplas, 1997; Lane, 1955; Galay, 1983). Aggradation typically results in an eventual increase of stream slope, all other parameters remaining the same. Long-term results of aggradation can include a gradual decrease in mean particle diameter, and an ability of the stream to transport larger particles due to an increase in energy slope (Diplas, 1997). Predicting the consequences upon slope long-term may be a useful design tool in this regard. Like a dam, a single perturbation or action may have one impact upstream (aggradation), and a different impact downstream (degradation) (Diplas, 1997; Dunne *et al*, 1978; Lane, 1955; Leopold *et al*, 1964).

Typical tributary loading characteristics with high sediment discharges include a deposit of materials at the confluence and aggradation progressing upstream. Downstream aggradation will also be caused by a cut-off of a meander bend or general reduction in sinuosity through a reach (Diplas, 1997; Dunne *et al*, 1978; Rosgen, 1996; Galay, 1983; Yang, 1996).

As another example of aggradation, at the turn of this century, significant changes in California streams were observed with an increase of material deposition associated with hydraulic mining of gold from the placer deposits. Using low-water level records at key downstream riverports, up to 10-15 feet of aggradation was observed in channels due to surface mining spoil wash-off. This was noted to co-occur with the increased large-scale disturbance of land introduced by the more advanced hydraulic mining technique. The mining in turn generated monumental increases in sediment flow, but did not compensate the system with increased water flow (Leopold *et al*, 1964). The result was aggradation.

Rates of aggradation in the example noted above ranged 0.25-0.33 feet per year over broad areas of impact. Similar but slower rates of aggradation may be associated with agricultural activities. In each case rates will depend primarily on the physical properties of the aggraded material and the streams natural capacity to carry away those materials.

2.14.10.3 Degradation

Counterpart to aggradation, degradation or scour is the removal of channel material and lowering of the channel bed. Degradation results in reduced energy slope at the location of scour. It can be expressed downstream of dams due to the release of clear water or due to a diversion of clear water into the basin (e.g. groundwater for instance) (Galay, 1983). Upstream degradation will also be caused by cut-off of a meander bend or general reduction in sinuosity through a reach. A drop in reservoir level can also cause degradation. Consequences of degradation can include (Diplas, 1997; Galay, 1983):

- Exposure of pipeline crossings and bridge piers,
- Water-levels can drop below water intake lines,
- Lowered groundwater table,
- Loss of land due to higher banks and bank failure, and
- Loss of washload to the floodplain.

2.14.11 Bank Instability

It has been shown that a natural stable stream channel is not straight, but meanders almost once every 10 to 20 stream widths (Diplas, 1997). It is best to understand that a stream is dynamic, and is always changing. Plan form geometry is constantly adjusting to balance inputs of sediment and water. Of notable importance, bank instability appears to be associated with what happens at the banks itself (Diplas, 1997), and can be perturbed by what happens in the channel.

Cited before, denudation of the hillside landscape may be occurring on the magnitude of removal of 1” to 2” of surface material every 1000 years (Leopold *et al*, 1964). Changes in channel plan form occur much faster than changes to the hillside because the valley floor reacts appreciably faster to changes in climate, relief, and vegetative cover. This is due to the concentrating effect of water’s energy (Leopold *et al*, 1964; Yang, 1996). To prevent lateral movement via channelization may actually increase channel instability.

2.14.11.1 Channel Bank Erosion

Channel bank erosion can be a significant source of sediment flow over short and longer periods of time. It can be of considerable importance in systems like Leading Creek that have highly erodible bank materials (Diplas, 1997; Lucht *et al*, 1985). For our time frames, this can really only be evaluated in the context of the degree to which associated plan form geometry is stable in the given climate, all other variables remaining the same (Diplas, 1997; Dunne *et al*, 1978; Leopold *et al*, 1964; Schumm, 1977). Streams move across valley floors all of the time. It is more a question of how fast is it happening, not is it happening.

In the end, what is causing the observed changes (changes in land use and cover), and acceleration of instability (positive or negative) appear to be the most important aspects to consider and manipulate (Bedient *et al*, 1992; Bicknell *et al*, 1993; Donigian *et al*, 1995; Richards *et al*, 1994b). Without explanation from changes in independent variables listed in Tables 2.4 and 2.5, anthropogenic induced erosion is likely activated by land surface induced perturbations (expressed in-channel). These perturbations are initially caused by changes in vegetation type and density or topographic relief (Richards *et al*, 1994b, 1996; Sweeney, 1993; USDA, 1983, 1985, 1998). The perturbation will eventually either be attenuated or amplified, causing significant change (Diplas, 1997).

As an example, Sweeney (1993) reported that the presence or absence of trees on land adjacent to stream channels is shown to significantly affect the structure and function of macroinvertebrate communities in White Clay Creek, a Piedmont stream in southwestern Pennsylvania. Sweeney indicated low order forested tributaries are about 2.5 times wider than deforested streams and have more benthic surface area in the form of inorganic (sand, gravel, cobble) and organic (tree-roots, leaf litter, wood, etc.) substrates for macroinvertebrate colonization (Sweeney *et al*, 1993). For the Leading Creek Watershed, it is unlikely that widespread bank instability observed today is the result of perturbations in transport that actual originated from inside the stream channel (i.e. the channel/valley morphological units). General deforestation and decline from pristine conditions dates to upwards of a 100 to 1000 years ago, with greater acceleration of deforestation and erosion occurring in the last 100 years (Dunne *et al*, 1978; Leopold *et al*, 1964; Sweeney, 1993).

These effects together with farming and mining have radically changed the channel structure in Leading Creek in two significant directions. These effects have both led to an acceleration of erosion with adverse impacts realized by aquatic organisms. In just 100 years:

- ✓ There is no longer a mature, diverse stand of trees with thick root structures inhabiting the stream banks (Sweeney, 1993), and
- ✓ Sediment derived from floodplain-silt and upland-sand has come to dominate an otherwise cobble/gravel bed (OEPA, 1991a, 1991b; USDA, 1983, 1985).

Similar effects would not be expected due simply to subtle changes in climate, which do not appear to be significant anyway in the last few hundred years (Lucht *et al*, 1985). This is also supported by the fact that relief and vegetation, key variables in the idealized fluvial model, have been the most perturbed variables within the last 100 years in this watershed. There is substantial technical evidence of this conclusion seen in the stream system. The resultant widespread impact to relief and vegetation from these actions has been documented (USDA, 1985). Endemic erosion features observed throughout channel-valley units in the Leading Creek stream system identify ongoing moderate bank erosion, reduced channel carrying capacity, and time-variant sporadic pulsing of large stores of fine sands or shifting sands; (Allan, 1995; Burton, 1992; Richards *et al*, 1994a, 1994b). These impacts are likely to be attributed to changes in hillside/drainage dynamics for this climate and time period (Schumm, 1977).

2.14.12 Riffle-Pool Configurations and Increased Sediment Loading

Engineering flume studies show that under shear-stress conditions below pavement mobility increases in fines loading accumulate to varying degrees in gravel beds (Diplas *et al*, 1992b, 1994). It has also been established that mobile pavement conditions are needed to clean fines from both the pavement and subpavement zones of gravel bars. Pool gravel will first begin to fill with fines, moving eventually towards the bar and its various distinguishable parts (e.g. bar head, bar tail, etc) (Diplas *et al*, 1992b, 1994).

Pools will be finer in sediment size distribution than areas of the reach associated with bar elements (Diplas *et al*, 1992b, 1994). Under increased fines loading, increased settling occurs throughout the reach and affects each structural element of the riffle-pool configuration slightly differently. Under continued depositional conditions, the inter-gravel framework will often experience a skin effect (or inner seal within the subpavement). Under continued loading without sufficient shear stress conditions, at some point the riffle bar will become buried by the fine material (Diplas *et al*, 1992b, 1994; Lisle, 1989). Excessive loading of coarser sand materials >0.125mm can also inundate gravel bed reaches. In Leading Creek for example, sands from mining have radically reduced the D_{50} of bed material, pavement, and subpavement, and have greatly altered physical habitat.

The subpavement skin effect, representing a horizontal barrier in the gravel bed to further fine material infiltration (<0.125mm), is considered to be roughly $2.5*D_{90}$. The seal caused by particle bridging is associated with the presence of fines (<0.125 mm) that have worked their way through the "viscous layer" down into the gravel bar itself. Described earlier, aside from fines loading, the make-up of inter-gravel particle size distribution does not otherwise change remarkably (Diplas *et al*, 1992b, 1994; Lisle, 1989; Parker *et al*, 1982; Vanoni, 1977). For example in Leading Creek, this indicates excessive siltation can change the entire character of subpavement or, with continued loading, pavement material. Hidden subpavement can be choked with silt and the pavement free of obvious siltation problems.

Just below critical shear stresses for the pavement, bed materials >0.125mm behave as bed load and suspended load, where these larger size particles bridge before infiltrating further down into the subpavement zones. The skin effect describes the point at which bridging occurs for fine materials like silt and clay. The main structure of inter-gravel framework (materials >0.125mm) is therefore greatly determined by sediment size distributions *in situ* at the time the hydrograph recedes below critical flow conditions for a given cross-section description (Diplas, 1992b, 1994; Lisle, 1989).

2.14.13 Riffle-Pool Fine Sediment Cycle

Summarizing sedimentation aspects of riffle-pool configurations (Diplas *et al*, 1992b, 1994; Lisle, 1989; Milhous, 1982):

- ◊ Given increased loading from upland hydrologic units, less than near-bankfull events will likely tend to fill pools and riffle-bar inter-gravel framework with fine sands, silt and clay (materials <0.125mm).

- ◊ Pool gravel will first begin to fill with fines, moving eventually towards the bar and its various distinguishable parts (e.g. bar head, bar tail, etc).

- ◊ The subpavement zone will experience a skin effect. Continued loading, without bed mobilization, will eventually inundate the subpavement first and then the pavement.

- ◊ Shear stress conditions above the pavement zone (but smaller in value than the pavement mobility threshold) will determine the degree to which the pavement itself is inundated by fines during lower flow conditions.

◊ Increases in concentration generally lead to increases in fine material found in the upper portions of the subpavement and pavement (Diplas, 1994).

◊ The subpavement make-up, due to increased washload infiltration, in general is more dependent on concentration of washload. It also appears to be relatively independent of sub-critical flow conditions, except for defining possibly final subpavement skin depth and the degree to which "packing" occurs during these lower flow regimes.

◊ Mobile pavement conditions are needed to clean fines build-up from both the pavement and subpavement zones of gravel bars.

For a given time period and climate, nature's alignment of individual riffle bars has been optimized. This optimization in some way is reflected by the particle size distributions and climate dictating near-bankfull flood recurrence intervals that determine clean-outs (e.g. 1-2 year clean-out cycle for fines) (Diplas *et al*, 1992b, 1994; Milhous, 1982). Regarding packing of the pavement by water overflow (Lisle, 1989), the longer the duration between near-bankfull storms, and the greater the magnitude of average sub-critical flows experienced, gives rise to "tighter" pavement and higher initial threshold shear stress conditions. This would presumably hold for each particle size fraction (Diplas *et al*, 1987, 1992b, 1994; Einstein, 1950; Parker *et al*, 1982).

Many riffle-pool sediment transport phenomena and underlying physical mechanisms described here are ignored or not adequately addressed in most hydrologic simulation models because of shear complexity. For these reasons, successful hydrologic simulation of washload in watersheds is a difficult task with models available today such as HSPF. Upstream gravel beds exist and dominate channel forms. The degree to which simulations are successful likely depends on the degree to which washload dynamics of those gravel beds are being captured by washload data sets at sand beds downstream. Current models, by default of calibration approaches used, are integrating gravel beds in ways that may not be physically meaningful under a wide variety of climatic and land use conditions that can span a thirty-year period. While this does not preclude their use, it underscores that uncertainty may be poorly described.

2.14.14 AML Sand Cycle

An exceptionally wet year in the Leading Creek regional area occurred in 1990 with rainfall 35-40% above normal mean annual levels (OEPA, 1991a). As a result, the 1991 OEPA Biological and Water Quality Survey impacts of AML noted correspondingly increased sediment transport and downstream bank instability (OEPA 1991a, 1991b).

Aspects of the following mechanism summary have been observed and reported in various studies of regional stream systems suffering the impacts of AML sedimentation. Historical studies included the Leading Creek Watershed (OEPA 1991a, 1991b; Leopold *et al*, 1964). The suspended sediment phase of AML (the coarser sand portion) relates a cycle that is predictable, appears to consistently prevail throughout eastern Ohio, and directly prevents widespread use-attainment in many reaches of Ohio's river's and streams, notwithstanding the added effects of acid mine drainage. The cycle conjectured follows a chronological path:

(1). Accelerated upland erosion and sedimentation occurs,
(2). Increased delivery of fines and sands to the channel follows,
(3). Fines and sands will eventually dominate local particle size characteristics,
(4). Excessive sediment bed loads develop,
(5). Progressive upstream aggradation builds resulting in increased slopes downstream, and decreased slopes upstream,
(6). Increased stream velocities at higher flows then occur due to higher slopes,
(7). Due to induced higher shear stresses, increases in the overall size of particles transported can increase,
(8). Increased shear stresses are exerted on bed and bank structures downstream system, and, finally,
(9). Destabilized banks develop in some areas, particularly those already under increased stress from denuded riparian zones in the floodplain.

OEPA studied and recorded observations at 30 watersheds that covered 62 separate reaches with 23 of those sites sampled along a single mainstem, the Hocking River. These streams all have some varying degree of AML impact. The above outline parodies in many ways the mainstem themes of impact OEPA recorded. Looking at these southeastern Ohio watersheds, OEPA observed that resultant channel-valley responses to increased sediment stress tended to be most severe at lower elevations of historical watershed relief.

2.14.15 Sediment Mechanics Summary For Leading Creek

The outcome of AML/AMD will of course depends upon the spatial influence that geology, relief, and stream mechanics (e.g. slope, D_{50}, water flow) combine to exert in each occurrence (Leopold *et al*, 1964; Vanoni, 1977; Yang, 1996). The AML sand cycle outlined above is expected to occur repeatedly throughout the Western Allegheny Plateau (WAP) ecoregion in areas with serious landscape disturbance, and having a consistent sedimentary geology of the southeastern Ohio coal belt. Agricultural and urban siltation is also widespread in these regions (Yoder *et al*, 1996).

Agricultural and urban siltation will occur and manifest as washload or turbidity in the water column, leading to increased fine material in bed sediments. For gravel bed streams, siltation will lead to a clogging of the inter-gravel framework, to a vertical depth defined by a seal thickness of 2.5 to 5 $*D_{90}$ of the subpavement material. This occurs first in pools and then in riffle bar areas. Siltation eventually leads to inundation of the subpavement and pavement materials.

A cycle of aggradation and degradation for mining sediments is consistent with Lane's equation and the idealized fluvial model that predicts a trend to stabilize any two-phase flow (Lane, 1955; Rosgen, 1996; Galay, 1983). Typical tributary loading characteristics with high sediment discharges will lead a deposit of materials at the mainstem confluence and aggradation progression upstream (Diplas, 1997; Lane, 1955; Yang, 1996). Stability in this case is an increase in meandering in response to increased channel slopes. With a trend to a more gentle slope upstream, an ever more formidable slope downstream will develop, which results in instability downstream. Energy, stress and resistance will attempt to then re-stabilize this new downstream configuration. A cycle ensues that affects areas both upstream and downstream of a given perturbation (Diplas, 1997; Lane, 1955; Leopold *et al*, 1964; Rosgen, 1996).

2.15 ROLE OF PARTICLE SIZE IN STREAM MECHANICS AND ECOLOGY

2.15.1 Importance of Particle Size Data

The importance of adequate measurement of particle size in gravel-bed streams is found in the widespread use of particle size in various engineering problems. These relate to description of bed materials and description of transport of those materials downstream with the flow of water. Key sediment transport mechanisms must therefore be evaluated over various particle size ranges accurately. As an example, the estimators D_{35}, D_{50}, D_{65}, and D_{90} are critical parameter values related to resistance and water flow, and incipient motion criteria governing sediment flow (Vanoni, 1977; Yang, 1996).

There can be inherent biases involved in conducting areal sampling of pavement, sub-pavement, and parent material in gravel beds. Biases can be introduced when converting these estimates to assumed volumetric equivalents for comparison between techniques, or comparisons over time within technique. Several issues are involved in controlling error associated with particle size distribution analysis of gravel beds (Diplas *et al*, 1992a, 1997; Fripp *et al*, 1993). The associated particle diameter statistics developed from such estimates will be themselves highly dependent upon the chosen technique to sample *in situ* and the number of samples collected (Church *et al*, 1987; Diplas *et al*, 1992a; Fripp *et al*, 1993; Kellerhals *et al*, 1971; Leopold *et al*, 1964; Wolman, 1954)

Effects of operator error introduced in the accurate estimation of particle size distributions of gravel bed pavement and subpavement can have far–reaching impacts upon engineering assessment and design (Vanoni, 1977; Yang, 1996). Impacted will be both estimates of the flow of water and the flow of sediment. With the value of ecology established (Suter, 1998a; USEPA, 1998; USDA, 1998), equivalent arguments can be made that accurate assessment of particle size is also of equal importance to assessment strategies in ecological risk assessment (Diplas *et al*, 1994). Impact to biodiversity and overall biotic integrity is intimately related to sediment characteristics (Burton, 1992; OEPA, 1989; Rankin, 1989; Richards *et al*, 1994a, 1994b; USEPA, 1989, 1996c).

2.15.2 Tracking Fine Materials in Gravel Beds

The intersecting needs of particle size data in River Mechanics, and the ecological importance of tracking very fine materials into and out of gravel-bed framework systems has been shown (Church *et al*, 1987; Diplas, 1992b, 1997; Lisle, 1989). Increasingly valued ecology will demand more accurate knowledge of operator error and correction of inherent test biases in particle size sampling. An objective of sound sampling is to establish volumetrically equivalent samples addressing four key areas (Diplas *et al*, 1992a, 1997; Fripp *et al*, 1993):

1. Representation of the true sediment populations,
2. Volumetric equivalence conversion,

3. Sample truncation, and
4. Sampling accuracy.

2.15.3 Ecological Significance of Riffle Bars

From a viewpoint of hydraulics, given increased loading from upland hydrologic units, less than near-bankfull events will tend to fill (and then cover) pool and riffle-bar inter-gravel framework with very fine sands, silt and clay (Diplas, 1994). As examples of the phenomena in Leading Creek, many pool/riffle bar units in urbanized and agricultural areas are inundated with fine clay/silt materials. More radical inundation by fine, medium, and coarse sands in Leading Creek also characterizes lower mainstem and tributary stems impacted by strip mining. Agricultural and urban siltation of the gravel framework, and the presence of the often deep AML sands that bury gravel beds, may have the following adverse physical impacts upon biota (exclusive of toxicity) (Diplas, 1997; USDA, 1998; Waters, 1995):

- Reduced the range of stream celerity (e.g. swiftness or velocity),
- Physically altered habitat,
- Reduced bed-interflow and transfer of water and nutrients,
- Induced mechanical and chemical stresses upon eggs and insects,
- Increased mean pool temperatures,
- Reduced mean pool depths, and
- Inhibited oxygen, mass, and energy transfer processes.

The above factors result in overall reduced biotic production in pools and riffles (Adams *et al*, 1980; Allan, 1995; Bradley *et al*, 1991; Burton, 1992; Diplas *et al*, 1997; Lisle, 1989). Riffle-bars themselves are the most diverse, productive zones within aquatic stream habitat (Allan, 1995; Burton, 1992; Diplas, 1997; Milhous, 1982; OEPA, 1989; USEPA, 1989). While the pavement in a gravel-bed is of more interest hydraulically, the accurate assessment of subpavement siltation leads our basic concerns for biota (Diplas, 1997; Diplas *et al*, 1997).

2.15.4 Spatial Dependence of Mean Particle Size Diameter

Natural alluvial streams are classified broadly into two types of systems, sand-bed and gravel bed streams (Church *et al*, 1987; Vanoni, 1977; Yang, 1996). This is in part motivated by a typical natural feature of alluvial streams to express a bimodal distribution with respect to mean sediment particle diameter from station to station along a mainstem (Diplas, 1997). This defines a large-scale spatial distribution of D_{50}'s (50% finer) longitudinally on the mainstem..

The effective D_{50} particle diameter distribution going downstream will be expected to have overall trends of larger values upstream (boulder, cobble, or gravel-beds) and lower values downstream (sand-beds) (Dunne *et al*, 1978; Leopold *et al*, 1964). D_{50} will be sometimes sporadic (up/down) depending on tributary sediment load inputs, but the trend will prevail. The origin of largest size will depend on geology, relief, and climate (Leopold *et al*, 1964; Schumm, 1977). Sediment heterogeneity instream is related to shear stress concepts, particle size, energy slope, abrasion, and imparted sorting characteristics. A great deal of variation in particle size distribution is encountered, making particle size distribution measurement difficult.

2.15.4.1 Particle Size Transition on the Mainstem

A jump from gravel-beds to sand beds (i.e. D50>2mm jumping to <2mm) is often realized in natural systems. At some point downstream a transition will be expected to occur on the mainstem over a short distance (Diplas, 1997; Dunne *et al*, 1978; Leopold *et al*, 1964). At this section, the stream will transition from gravel-beds to sand-beds, in the channel-portion of the idealized fluvial model (e.g. hillside-channel-delta) (Schumm, 1977). D_{50}'s will typically not be recorded at the 2mm level. Thus the effect of this natural sorting process is to leave a gap in D_{50}'s across the fine-gravel/coarse-sand range, resulting in a bimodal distribution (Diplas, 1994; Lisle, 1989). Unimodal distributions in D_{50} can occur, depending on all environmental factors. What this implies is that there will likely also be an abrupt transition in successful techniques which can be used to actually measure mean particle diameter for a given stream bed (Church *et al*, 1987; Diplas *et al*, 1992a; Fripp *et al*, 1993, Diplas *et al*, 1997).

2.15.5 Sampling Particle Size in Gravel Beds Vs Sand-Beds

To summarize the differences between sand and gravel-beds, sand beds are adequately described by the associated D_{50}, where as D_{50} is not sufficient alone in describing gravel beds. Gravel beds require a full description of the particle size distribution, thus there is an importance in sampling all size fractions accurately. Gravel beds are also partitioned vertically into three horizons: *the bottom layer, sub-pavement, and pavement zones*. Only the bottom layer is amenable to bulk sampling, and only if a large enough sample can be collected (Diplas, 1992b, 1994; Lisle, 1989). As discussed further below, areal samples are biased volumetrically, and include truncation.

Researchers have used liquid CO_2 and/or cans to collect (frozen) samples of pavement and subpavement successfully (Adams *et al*, 1980; Lisle, 1989), but this can be difficult, and is limited by particle size. For any given gravel-bed reach, it can be recognized that the task is often accurately sampling particles over the entire range of sediment encountered to develop various estimators. The sampled sediment particle population can range from perhaps boulders a few meters in diameter down to fine sands < 0.125mm, to silts < 0.075mm, down to clays < 0.002mm that are always present (Klingeman *et al*, 1993; Vanoni, 1977; Yang, 1996).

For sand-beds, particle size is relatively easily and accurately measured using bulk sampling methods (e.g. a jar, shovel or sampling dredge or other technique as needed). For any bed, if silt/clay size distribution or chemical analysis is critical, extreme care should be taken to not disturb, discard, or displace precipitation layers (like acid mine drainage flocculants) or pore water during sampling (Babendreier *et al*, 1997; Burton, 1992).

2.15.6 Truncation of Size Fractions in Sediment Samples

The typical approach to sampling substrates in gravel beds is via use of pebble counts collected in some manner, such as the Wolman Walk (Church *et al*, 1987; Kellerhals *et al*, 1971; Leopold *et al*, 1964; Wolman, 1954). Cobble or gravel bed sampling methods though ignore fine materials in the bed, effectively truncating these size ranges from the sample collected at about 15mm. Most attempts to exclude larger particle sizes in environmental samples (Adams *et al*, 1980; Burton *et al*, 1992) collected in the field also present inherent problems, such as sampling for sediment toxicity testing. While exclusion or bias in this manner can be immediately more meaningful and simpler, it might be better in the long-term if all data were collected and strictly analyzed by ASTM methods (ASTM, 1990b, 1993; Gee *et al*, 1986; Leopold *et al*, 1964). Data can always be combined or excluded later, but each size fraction or pebble/boulder measured should be properly recorded in the field and remain an inherent feature of the particle size distribution data set for a given stream section or bar form. Notwithstanding, ASTM methods need improvement to address biases in collecting field samples from gravel-bed systems.

After field sampling is completed, particle size analysis, for example using hydrometers or centrifuges are assumed to have similar difficulties in operator error, for example regarding sub-sampling of field samples. ASTM standards for particle size analysis, if followed, should prevent significant operator error issues throughout any subsequent sub-sample analysis (ASTM, 1990b, 1993).

Many ecological studies of gravel bed sediment, developing quantitative data on %sands, %silt, and %clay, often radically bias gravel bed distributions by (a) preferential sampling of the reach, and (b) standard exclusion of materials > 2mm (Babendreier *et al*, 1997). Thus the goals of ecologists and engineers may have vastly different concerns regarding gravel beds, and field-sampling plans must account for these differences, if any. Presentation of sediment particle size data needs to set out to what degree sediments being reported have been selectively biased.

Unfortunately, sediment particle size analyses collected in ecological risk assessments represent a great deal of work though much of the standard physical data collected can have limited utility to both ecological and engineering applications. Efficacy of data is reduced due to bias introduced in collecting quantitative samples, or for example due to the qualitative nature of Rapid Bioassessment Protocol (RBP) habitat data. For example, RBP substrate, embeddedness, and cover data is qualitative and arguably subjective (Rankin, 1989; USEPA, 1989). From both ecological and engineering aspects, the degree to which biasing particle size data currently limits our knowledge of landscape influences on biota is not well understood.

2.15.7 Bedload Sampling

Collection of accurate bed-load data with Helley-Smith (pressure-differential) samplers is difficult in large part because sediment bedload rates vary significantly under seemingly similar flow/head conditions. Extreme spatial and temporal variation in Helley-Smith sampled bedload transport rates has been shown (Glysson, 1993). This error is not expected due to operator error per se, but due to variation in actual transport rates occurring during measurement. The USGS paper offers the most up to date policy on bedload sampling with Helley-Smith samplers. Operator errors can also be induced by inadequate equipment selection, unacceptable field conditions for application of this method, and inappropriate sample composting (Glysson, 1993).

2.15.8 Temporal Variability of Stream Bed Composition

It has been shown (Adams *et al*, 1980) that the percentage of fine materials in gravel bed frameworks varies greatly with time. Therefore regarding concerns for particle size sampling and channel descriptions, it is also recommended that if one can only sample once, low flow conditions are the best opportunity, when beds are stable (Adams *et al*, 1980). The temporal variability is due to the net flushing of fines; driven by near bankfull shear conditions that provide for subpavement mobilization, on occasion. This recommendation also coincides with field conditions that make it easiest to reduce operator error associated with hybrid sampling (Diplas *et al*, 1997).

Effects of subpavement seal depth and related concepts of bed packing can therefore also bias time-sequenced measurements of sediment particle size distribution (Adams *et al*, 1980; Lisle, 1989; Diplas, 1994). Sampling plans used to monitor in bed composition must attempt to characterize seasonal changes in bed composition (Adams *et al*, 1980; Diplas, 1997). It would appear that some ability to describe hydrologic-hydraulic conditions for several months up to bed unbiased gravel-bed sampling would be helpful in describing potential variation induced by infiltration of fines and bed packing phenomena.

Temporal fluctuations can be large enough to obscure effects of changes in land-use determined via bed composition analysis (Adams *et al*, 1980). Adams and Lisle seemed to indicate that the beds can stabilize quickly with skin/seal effects set in perhaps 15-30 days or shorter. Their research also showed the actual pavement and subpavement framework material (particles >0.125mm) stabilize relatively instantaneously, once water-depth falls well below incipient motion conditions, for any given fraction (Diplas *et al*, 1992b, 1994; Lisle, 1989).

Mounting evidence exists indicating that land-uses do contribute to cyclical siltation of the gravel-framework, choking gravel-bed health (Adams *et al*, 1980; Allan, 1995; Diplas 1994; Lisle, 1989; Milhous, 1982). For this reason, measuring any changes associated with the clogging effect of gravel-beds can be critical and insightful in evaluating the success of stream restoration efforts aimed at reducing impacts of mining, forestry, and agriculture sedimentation upon biota.

2.15.9 Hybrid Particle Size Sampling Procedure

Introduced above, the main goals of sediment sample collection and analysis in estimating particle size fractions is to address (1) population, (2) volumetric conversion, (3) truncation, and (4) sample accuracy. Among grid by number, areal, and volumetric samples that can be collected, the volumetric sample is the only unbiased sample (Diplas *et al*, 1997; Kellerhals *et al*, 1971). Grid-by-number samples are volumetric equivalents, but areal samples must be converted (Kellerhals *et al*, 1971).

2.15.9.1 Sample Population

The following summarizes key information provided concerning the hybrid sediment sampling procedure (Diplas *et al*, 1992a, 1997; Fripp *et al*, 1993). The procedure combines the typical grid by number pebble counts (Church *et al*, 1987; Leopold *et al*, 1964; Wolman, 1954) conducted to ascertain the particle size of larger fractions of a gravel-bed >15mm. Added to the pebble count procedure, an adhesive areal sample is also collected to resolve the problems of truncation of smaller pavement materials that occurs with the pebble count. The adhesive sample characterizes data < 40mm. A match point is then used to combine the distributions of both samples to arrive at a single size distribution (Diplas *et al*, 1997).

The pavement layer is approximately D_{90} thick, and the subpavement 2 to 2.5*D_{90} thick (Diplas, 1992b). Thus a volumetric (bulk) sample cannot be directly sampled for the pavement and subpavement, as can the bottom layer (Church *et al*, 1987; Diplas *et al*, 1997; Kellerhals *et al*, 1971). A direct volumetric sample of the pavements would end up mixing the various, distinct pavement, subpavement, and bottom layers zones of the bed. The adhesive procedure appears to be applicable for use on subpavement sample populations once a pavement sample is similarly removed from the top, provided particle sizes are small enough and is amenable to aereal sampling (Diplas, 1994). Sampling materials of subpavement when D_{90} is > 40 mm would though appear problematic in determining coarser fractions. It is emphasized though that smaller fractions of the subpavement are of most interest ecologically where development of an unbiased procedure to accurately track embedded fines would be of greatest practical interest.

2.15.9.2 Sample Truncation and Volumetric Equivalence

For most gravel-beds, the courser materials of the pavement layer can be sampled using a grid by number method like the pacing pebble count method (Church *et al*, 1987; Leopold *et al*, 1964; Wolman, 1954). This might include use of a gravelometer for smaller substrates, but truncation is still involved for particles less than an index finger, or ~15mm (Diplas *et al*, 1992a, Fripp *et al*, 1993). This truncation is a logically accepted limitation imposed by use of the hand and big toe in paced pebble counts. This truncation can lead to underestimation of the percentage of the gravel bed comprised of finer materials < 15mm. The basic approach of using the adhesive is to collect a representative sample of the smaller fractions, where stones > 40 mm will be truncated. The samples are then matched, after generating individual size distributions for each range. If large cobbles/boulders are present, a tape measure may actually be needed to complete the grid by number assessment of the larger size fraction (e.g. boulders >216mm) (Church *et al*, 1987; Diplas *et al*, 1997).

Having addressed the first three key ideas of unbiased sample collection, the area-by-weight samples are converted to volumetric equivalents using a formula developed by Kellerhals and Bray (Diplas *et al*, 1997; Kellerhals *et al*, 1971). A parameter estimate for the exponent of particle size geometric mean, specifically addressing the clay adhesive materials used, was also developed (Diplas *et al*, 1997). To address truncation aspects, the underestimated pebble counts, and overestimated adhesive samples are balanced using the match point approach. A match point is selected graphically that can effectively integrate the two distributions to arrive at an unbiased estimate of true *in situ* particle size distribution.

2.15.9.3 Particle Size Distribution Sample Accuracy

The fourth key of accurate particle size distribution estimation is the address of sample accuracy. Sample accuracy was well discussed by Fripp and Diplas (Diplas *et al*, 1992a, 1997; Fripp *et al*, 1993), where the use of a statistical approach to estimate minimum sample size for grid number samples was described based on a desired accuracy level. This method overcame other previous approaches available at the time, which often depended on having detailed knowledge of the distribution before sampling. To establish a certain level of sample accuracy, the idea is to know how many stones are needed in a pebble count walk or how much area must sampled in an adhesive approach (Diplas *et al*, 1997).

Accuracy for adhesive areal sampling has been previously delineated (Diplas *et al*, 1992). The work provided a stated level of accuracy based on the number of adhesive samples to collect per sample (e.g. per bed or reach site sampled). The basic assumptions made were that:

a) On each sediment surface, each grain on average projects an area proportional to the square of its sieve diameter.

b) A size fraction occupying p% of the volume of solids will occupy p% of the sample area occupied by grains.

With two assumptions, assuming n is the number needed for an accurate grid by number approach, the corresponding minimum surface area of an areal sample of the same accuracy is $n*D_m^2$, where D_m is the largest particle diameter to be sampled. With the hybrid sampling method, operator error should be kept to a minimum, and a desired level of accuracy can be determined, prior to sampling. This is one of the few approaches that can be used where an unbiased, volumetrically equivalent sample can be collected and documented in such a manner.

2.16 AG AND AML STRESSOR MECHANISM SUMMARY

As indicated by USDA, the following citation summarizes the inherent perturbation and system response attributed to agricultural and mining disturbances in a given watershed region and stream corridor (USDA, 1998):

> *"Streams and stream corridors evolve in concert with and in response to surrounding ecosystems. Changes within a surrounding ecosystem (e.g., watershed) will impact the physical, chemical, and biological processes occurring within a stream corridor. Stream systems normally function within natural ranges of flow, sediment movement, temperature, and other variables, in what is termed "dynamic equilibrium." When changes in these variables go beyond their natural ranges, dynamic equilibrium may be lost, often resulting in adjustments in the ecosystem that might conflict with societal needs. In some circumstances, a new dynamic equilibrium may eventually develop, but the time frames in which this happens can be lengthy, and the changes necessary to achieve this new balance significant.*

Any disruption of the dynamic equilibrium is of great interest as it represents a potential deviation from natural system response and biotic condition. In the eastern United States, for example, researchers have reported the true nature of the bountiful forested equilibrium prior to disturbance (Sweeney, 1993). Agriculture (AG) and Abandoned Mined Lands (AML) represent two major source classes of disruption in eastern United States watersheds. While both AG and AML sediments can physically inundate habitat, and carry adsorbed metals and organic material, AMD associated acid-base chemistry is relatively more complex (Nordstrom, 1982; Snoeyink *et al*, 1980; USEPA, 1980). AMD progression includes dissolution of pyretic materials, formation of sulfuric acid, and oxidation/reduction, chemical neutralization, and precipitation of various iron, manganese, and aluminum compounds (USEPA, 1976a, 1980).

Briefly summarizing processes involved in AMD generation, local acidification and metal toxicity tend to act in concert and are subject to the laws of open carbonate-system acid-base chemistry. Contaminants partition across various media; gas-air, solid-sediment, and liquid-water (Fetter, 1993; Snoeyink *et al*, 1980). From stream mechanics, sand transport is stream (carrying) capacity limited, which defines the magnitude and boundaries of its current impact. Adverse impact from AML sedimentation and the presence of contaminants due to AMD in Leading Creek is of course relative to their background concentrations.

Heavy metals including arsenic, copper nickel, and zinc can be associated with AML sediments, and when bioavailable, are toxic to indicator organisms used in controlled toxicity tests (Belanger *et al*, 1986; Mac *et al*, 1984; Pratt, 1990). These impacts can manifest structural and functional responses to introduced pollution, for example Schultheis and others indicate copper pollution significantly interrupts the actions of shredders processing leaf materials (Schultheis *et al*, 1997).

2.16.1 AML/AMD Mechanisms

A great deal at this point is known about AML, but there remain many questions on related key mechanisms (Daniels *et al*, 1995; USDA, 1983, 1985, 1998). Examples of ongoing discussion of AML-AMD impacts include evaluation of kinetics of microbial mediated oxidation (Bigham *et al*, 1996; Mac *et al*, 1984; Nordstrom, 1982; Rimstidt, 1997; Snoeyink *et al*, 1980; Williamson *et al*, 1994). Another area of current interest is analysis and modeling of acid generation with complex solute transport of heavy metals and earth minerals on relatively large scales (Birge *et al*, 1995; Broshears *et al*, 1996; Boult *et al*, 1984; Cherry *et al*, 1995, 1999; Hedin *et al*, 1994; Klapper *et al*, 1995; LaPerriere *et al*, 1983; Moloney, 1993; OEPA, 1994; Scannell, 1988; Schnoor *et al*, 1987; Stark *et al*, 1994).

Aspects of sedimentation from mining and associated physical properties of AML sediment have also been studied (Bonta *et al*, 1992; Cherry *et al*, 1979a, 1979b; Curtis, 1973, 1979; Dickens *et al*, 1989; Gray, 1996; Dunne *et al*, 1978; Higgins *et al*, 1987; LaPerriere *et al*, 1983; Leopold *et al*, 1964; Stewart *et al*, 1992). What is known about AML and AMD points to expectations that AMD impacts can be severe and widespread, and are usually complicated in their expression in the environment. Impacts of AMD are in large part controlled by pH (USEPA, 1976a, 1980). It is known that adverse impacts are expensive to resolve, if not prohibitive. In one natural remediation approach, application of passive wetlands treatment has been found to be usually limited to small design flows (Daniels *et al*, 1995; USDA, 1985, 1998).

The long-term discharge of acid leachate from AML spoils and excavation zones are major problems where eventually oxidation of pyrite will continue to completion (Daniels *et al*, 1995, Snoeyink *et al*, 1980). This aspect presents a more challenging field condition than does sedimentation and surface revegetation needs. Address of acidification of mine spoils usually entails some scheme of neutralization, and might require interbedded bulk-blending of alkaline materials such as ground agriculture limestone up to mixture ratios of 5%, greatly increasing costs of reclamation (Daniels *et al*, 1995). AMD reclamation is most efficiently done at the time of waste pile construction, which is not economically feasible for most historic AML sites with AMD impact. Direct study of impacts of AML upon ecology that differentiate sedimentation and AMD are limited (Daniels *et al*, 1995).

2.16.2 Corridor Condition and Sediment Loading

Control of sedimentation has three phases (Waters, 1995):

- **Prevention** - arresting original erosion or obstructing eroded sediment from leaving the site of its origin,

- **Interdiction** - capturing and retaining sediment before it reaches the stream (i.e. sediment traps), and

- **Restoration** - removing sediment from the stream to bring physical conditions back to their original state.

Sediment load in natural streams is a function of the cumulative disturbance within the watershed. Activities that promote the potential for erosion, i.e. the removal of vegetation and alteration of landscapes, will increase sediment-yield (Curtis, 1973, 1979; Dickens *et al*, 1989; Olyphant *et al*, 1991; USDA, 1998; Waters, 1995). Clogging and in some cases saturation of inter-gravel matrix voids with silt and clay fines can occur in gravel beds (i.e. riffle bars) reaching from the headwaters to various areas of the mainstem. For example, at RKM 16 on Leading Creek, the gravel-bed nature (Church *et al*, 1987) of the stream transitions where the stream takes on a more consistent and smaller D_{50}, in the fine sand range.

The effect of gravel bed siltation further upstream of the sand bed regions results often in an effective blockage of water transport, nutrients, and oxygen through the gravel pore spaces (Diplas, 1992b). This level of inter-gravel matrix siltation directly inhibits reproduction cycles, from both a physical and chemical standpoint (e.g. crushed fish eggs, decreased dissolved oxygen, inter-gravel flow of water and nutrients, etc.) (Bradley *et al*, 1991; Berkman *et al*, 1987; Diplas, 1994; Diplas *et al*, 1992b; Lenat, 1984; Lisle, 1989; Lloyd, 1987; Luedtke *et al*, 1976; Miller, 1988; Nelson *et al*, 1992; Richards *et al*, 1994a).

Riffle zones are the most productive habitat in a weakly structured pool-riffle stream sequence (Allan, 1995; Diplas *et al*, 1992b; Richards *et al*, 1994b, Richards *et al*, 1992; Rosenberg *et al*, 1993; Yoder *et al*, 1996). Habitat for both fish and macroinvertebrate therefore can be radically disturbed by the “physical” intrusion of sedimentation into gravel beds. Other mentioned aspects of sedimentation are also potentially severe including turbid water, reduced photosynthesis and impacts to energy production/availability.

Reduced biotic production capacity associated with an otherwise potentially diverse aquatic environment will result from sedimentation. Across Leading Creek, overall flood plain encroachment and channelization of the stream corridor are other obvious contemporary deviations from historical potentials of this stream system for this time period and climate.

2.16.3 Differentiating Agricultural and Mining Sedimentation

In many watersheds impacted by agricultural (AG) run-off, pre-1977 coal extraction and associated acid mine drainage (AMD) can also play a key role in defining water and sediment quality (OEPA, 1991a, 1991b; USDA, 1985). Downstream chemical and sediment properties in turn are increasingly being used to sample and diagnose the watershed holistically (Cairns *et al*, 1991; USEPA, 1996c).

Future restoration alternatives will tend to be selected via watershed-scale analysis, and then implemented at individual field-scales. A requisite task remains to differentiate significant field-scale sources of AG and AMD stress to better relate cause and effects of various AG and AML/AMD management techniques. Since AG and AML/AMD can individually and jointly influence sediment and water quality, proper risk assessment must account for spatial and temporal distributions of both source classes under various management schemes (USEPA, 1996c). At the same time, each major class of non-point source pollution may have discriminant features (e.g. ASML versus AUML) in addition to concerns presented by urbanization.

Urban, AG, and AML/AMD's unique origins (flood plain and upland soils, respectively) make it feasible to distinguish AG and AML sediment run-off (Gillmore *et al*, 1991; Lucht *et al*, 1985). It is conceivable AG and AML/AMD will characteristically provide unique relative physical and chemical signatures, both upon reaching and then transporting associated pollutants down the receiving stream system. This was the initial experience found during reconnaissance of water column pollutants, depressed pH, and gross trends of particle size in Leading Creek.

To date, signature aspects of AMD and agriculture do not appear to have been readily developed or exploited in many quantitative approaches to risk assessment. In the same light, nor has the impact of co-occurring sub-classes of AG/AMD been studied in depth to relate physical habitat and biodiversity metrics used by ecologists (Daniels *et al*, 1995; Karr *et al*, 1981a, 1981b, 1985, Lenat, 1993; Merrit, 1984; OEPA, 1989; Yoder *et al*, 1996). Metrics to be used in this study should be able to differentiate dominant adverse impacts of urban, AG and AMD presented within Leading Creek. This was the thrust of work investigated here, placed in context of relating land use/cover to habitat, toxicity, and biodiversity.

2.16.3.1 Properties of Agricultural and Mining Sediment

As an example of physical properties of AML refuse piles from coal operations in southwest Virginia, the pile may have a median value of 60% materials > 2mm. Of the materials finer than 2mm, 60% of this may be sand, 22% silt, and 15% clays, resulting in sandy loam texture (Daniels *et al*, 1995).

In Leading Creek, over half the watershed is affected by abandoned mined land (AML) sedimentation or acid mine drainage (AMD) impact, and over half by agricultural sedimentation. Cobble/gravel bed characteristics would be expected to dominate many parts of the existing network system, as would silt loam characteristics of almost all hillside and floodplain soil series recorded in the area. Silt loam soil series dominate all landscapes except for the sandier soil series (Pinegrove; Pn) describing unreclaimed strip mines (Gillmore *et al*, 1991). As a typical example, the Taggart silt loam soil series found on some agricultural terraces in Little Leading Creek is defined by 90-100% material finer than 0.42 mm (medium sand), and 70-90% finer than 0.074 mm (silt) (Yang, 1996). Unreclaimed AML is characterized by 60%-85% materials coarser than silt, where 15%-60% is coarser than a medium sand.

While many aspects of AG run-off have been investigated (Bicknell *et al*, 1993; USEPA, 1991; Zison, 1980; Donigian *et al*, 1995), mining and reclamation activities are also known to significantly change watershed soils and vegetation, and increase sediment load (Bonta *et al*, 1992; USEPA 1976b, 1991; USDA, 1985). In many parts of the eastern United States, including Virginia, West Virginia, and Ohio, agriculture run-off and mining impact derived from pre-1977 activities represent two of the most influential and widespread classes of NPS pollution (Cherry *et al*, 1995, 1999; Smith *et al*, 1998; USEPA 1991; USDA, 1983, 1985, 1998; Waters, 1995). Spoil slides, haul roads and the mined area itself are three specific and major sources of sediment made available for transport into streams (Curtis, 1973; Dickens *et al*, 1985).

2.16.3.2 AML Sediment Profile

Thick sand deposits instream prevail in most of many AML-impacted tributaries to varying degrees (Gallagher *et al*, 1999). In Leading Creek silt and clay content in bed materials increase rapidly along the mainstem below Middleport, Ohio owing to the reduced bed load inputs, reduced water-surface slope, and backwater pool influences of the Ohio River (see Figure 2.4). The sandstone cap-rock materials were reportedly typically first removed during the surface contour mining operation with use of explosives, eventually discarded by force of gravity and wash-off downhill from the blast and extraction zones. Higher coal percentages and sulfide acidity potential are likely associated with increasing impact at mine gob and spoil areas derived from surface and underground mining. Other underground and surface mining operations generating residual surface spoils are included in the generic “AML” terminology used in this study, unless otherwise distinguished.

Excluding effects of bars and bed forms, bed sediment depths associated with contour mining operations are variable and can range several feet thick (>5 ft. to perhaps 10 ft. in some areas). Several personal accounts of historic stream conditions by local citizens in the Leading Creek Watershed and those nearby, indicated that AML sand deposition can literally reach bridge decks or otherwise completely fill stream channel profiles (upwards to 6 to 7+ feet deep). In areas outside of Leading Creek, stories were also recounted of streambed mounding in some cross-sections were sands piled to higher relief than the original loam banks. One can quickly realize the increases in slope that can occur in the near and far fields, directly below significant depositional areas, where aggradation slowly creeps upstream.

2.16.4 AG and AML/AMD Risk Assessment and Modeling

Conducting sound ecological risk assessments is a great challenge in today’s world where conceptually many ideas exist on how best to focus a watershed investigation (Black, 1997; Cairns *et al*, 1977, 1996; Doyle, 1997; Herricks *et al*, 1985; Kershner, 1997; Minshall *et al*, 1985; NRC, 1992; Renner, 1996; USEPA, 1992, 1998). Historically most approaches in ecological risk assessment first focused on chemical toxicity, and later on sediment toxicity in evaluating nonpoint source pollution impact to biota. In today's framework of risk assessment, macroinvertebrates have been found to be good overall ecological indicators of the stream system (Richards *et al*, 1992, 1994b; Rosenberg *et al*, 1993; USEPA, 1992, 1996c, 1998).

For agricultural sources, many field-scale and watershed-scale modeling approaches exist today which can, to some degree, adequately characterize sediment yield and expected instream metal/pesticide toxicity associated with primarily washload (Arnold *et al,* 1990, 1995; Bedient *et al*, 1992; Bicknell *et al*, 1993; Singh, 1995; USEPA, 1991). An analogy that relates the crucial relationship sought after is that no catchment outlet discharges in a vacuum exclusive of anything upstream effects. To attempt to relate landscape to water quality and flow is a common theme in watershed modeling (Bedient *et al*, 1992; Bicknell *et al*, 1993; Donigian *et al*, 1995; Linsley *et al*, 1975; Singh, 1995). One often finds themselves standing downstream and looking back upstream looking for answers to explain deteriorating ecology. Agriculture received a great deal of attention early on as a major nonpoint source of pollution to lakes and streams (Donigian *et al*, 1995; Fetter, 1993; Lenat, 1984; Osborne *et al*, 1988; Thomann *et al*, 1987; USEPA, 1976c, 1991; USDA, 1983; Zison, 1980). Less experience exists with AMD and mining sedimentation.

2.17 IMPACT OF AGRICULTURAL PRACTICES

Agricultural activities that have significant potential to present risks to ecology include (Anderson *et al*, 1976; USEPA 1976c, 1996; USDA, 1983; 1985):

- Non-irrigated crop production
- Irrigated crop production
- Rangeland
- Pastureland
- Feedlots
- Animal holding areas
- Animal operations

Within the Leading Creek watershed, the most influential agricultural practice is crop production, where the majority of crops receive no irrigation. Rangeland and pastureland are also common land uses in Leading Creek, especially in the upper half of the watershed (Cherry *et al*, 1996, 1997, 1999).

2.17.1 Agricultural/Urban Siltation Impacts

Agriculture and to a lessor degree small urban operations in the Leading Creek Watershed appear to have resulted in shallower pools, and have silted inter-gravel framework matrices of most gravel bars (riffles) along the mainstem. While much less dramatic in its effect upon biota than AMD, agricultural siltation is a key contributor to subtle ecological impacts limiting attainment of warmwater habitat (WWH) and exceptional warmwater habitat (EWH) by:

- ❖ Jeopardizing existing WWH status of the upper mainstem,
- ❖ Inhibiting most upper mainstem reaches from attaining EWH status, and
- ❖ Exasperating downstream sedimentation and AMD impacts derived from AML.

Many upper mainstem sites meet warmwater habitat (WWH) use-attainment status, but not exceptional warmwater habitat (EWH). The factors limiting EWH attainment in these reaches appear to be numerous and complex interactions of highly variable site specific stresses. With dominant anthropogenic influences in almost all reaches to be studied, it is not likely any one quantitative metric or small set of metrics could be derived to predict EWH score trends successfully at most sites in Leading Creek. This is particularly difficult to assess when no samples in the population meet EWH status (OEPA, 1994; USEPA, 1996c; Yoder *et al*, 1996).

It is believed EWH status is likely a resultant interaction between modest contributions from many parameters and mechanisms. Departure from EWH use-attainment is controlled somewhat by chemical, but also and perhaps more strongly by physical aspects of a reach. EWH would be expected to assimilate all subtle differences in local geology, run-off conditions, and derived water quality condition; floodplain, riparian zone, and bank encroachment; overall total upstream sedimentation (spanning silt and cobble particle size fractions), and dissolved load.

2.17.2 Agricultural Best Management Practices (BMPs) and Related Efforts

Best management practices that can potentially be used in Leading Creek tributaries include (Cherry *et al*, 1999; USDA, 1998):

- Livestock restriction from stream banks by fencing,
- Development of livestock stream crossings,
- Construction of alternative watering systems,
- Development of a riparian buffer zone between streams and fields,
- Stabilization of stream banks, and
- Cultivation of native riparian vegetation in the buffer zone.

Stabilization of stream banks includes a method of securing vegetation, providing a gradual, sloped bank, controlling bank erosion, and minimizing adjacent field erosion.

2.17.3 Integrated Watershed Management Schemes

In addressing nonpoint source pollution (USEPA, 1976c, 1991) to effect organized watershed restoration and holistic management, one may be eventually expected to include three focused methodologies in any successful approach (USDA, 1998):

(1) Stream Corridor Restoration,
(2) Best Management Practices for Agriculture, and
(3) Best Management Practices for Forestland.

A good deal of detailed research and experience is now being gained in the application of various restoration techniques. These techniques integrate and address many environmental aspects, including traditional engineering issues, and those couched in the framework of regulatory concerns (Bensch *et al*, 1998; Derrick, 1998). Several procedures define each of the three approaches noted (Cherry *et al*, 1999; USDA 1998).

2.17.3.1 Stream Corridor Restoration

- Brush mattresses
- Livestock exclusion or management
- Riparian forest buffers
- Riprap with joint vegetation
- Sediment basins
- Stone toe protection
- Stream meander restoration
- Tree cover

2.17.3.2 Best Management Practices: Agriculture

- Conservation tillage
- Contour farming
- Critical area planting
- Filter strips
- Integrated pest management
- Nutrient management
- Sediment basins
- Terracing
- Water storage management

2.17.3.3 Best Management Practices: Forestland

- Fire management
- Forest chemical management
- Forest wetland management
- Pre-harvest planning
- Revegetation of disturbed areas
- Road construction or reconstruction
- Road management
- Sedimentation controls
- Site preparation and forest generation
- Streamside management measures
- Timber harvest

2.17.4 Key Issues on Non-AML Best Management Practices (BMPs)

With respect to anticipated issues for Leading Creek, the following notes summarize key elements of various restoration strategies and techniques that could be eventually considered on Leading Creek (Cherry *et al*, 1999; Gore, 1985; Sweeney, 1993; USDA, 1998):

- Riparian buffer zones prevent the movement of sediment from fields and pastures into waterways. These buffer zones also protect the shoreline vegetation from erosion and sloughing.

- Shoreline vegetation keeps stream banks intact and stable, this becomes critically important when water levels reach bank full conditions under rain events.

- Streamside riparian forests affect food quality and quantity for macroinvertebrates directly through inputs of particulate food and indirectly by affecting the structure and productivity of the microbial food web.

- In agriculturally stressed ecosystems, the exclusion of livestock from creeks and streams is a crucial restoration technique.

- Tree cover and brush mattresses can be used to help stabilize banks where continued erosion from sediment loading, debris and floodwaters have removed valuable shoreline cover.

- Riprap is the procedure that provides the sturdiest shoreline protection and is often one of the most expensive approaches at a given site, and often the least ecologically favored.

Concerning riprap sturdiness, the analysis should account for natural stream response to channel restriction. Compared to some potentially more preferable modern-day bioengineering approaches, riprap may also not fully recognize undesirable consequences associated with channelization and/or inorganic stream bank protection (Derrick, 1998; Millar *et al*, 1998; Miller *et al*, 1998; Sotir, 1998).

2.18 FORMATION OF AMD

Detailed knowledge of sulfide mineral oxidation was described in literature decades ago along with the impacts of AMD attributed to mining hard rock ores (e.g. copper, gold, etc.) and coal (Dunne *et al*, 1978; Leopold *et al*, 1964; Nordstrom, 1982; Singer *et al*, 1970; Snoeyink *et al*, 1980). These earlier attempts are today still often referenced frameworks for describing the reactions resulting in acid mine drainage. An evolving issue, researchers began to recognize the important roles played by *Thiobacillus ferrooxidans* and other iron eating bacteria such as *Thiobacillus thiooxidans* and *Ferrobacillus ferrooxidans* (Daniels *et al*, 1995; Evangelou, 1995; Singer *et al*, 1970).

The acidophilic, mesophilic, microaerophilic bacteria represent rate limiting determinants in AMD kinetics in natural settings. An obligate chemoautotrophic aerobe, *T. ferrooxidans* dominates this process and has optimum growth in the range of pH 1.5 to 3.5 (Diz, 1997; Kuenen *et al*, 1992). Optimized relative activity of *T. ferrooxidans* has been reported near pH 3, 30 ^{o}C, and at O_2 molar fractions = 1% (Evangelou, 1995; Jaynes *et al*, 1984). Cell activity was reported as invariant to O_2 above 1% where pH and temperature represented local maxima. No significant growth of *T. ferrooxidans* occurs in saturated environments where O_2 diffusion is 10^{-4} times slower than diffusion in unsaturated waste pile systems (Evangelou, 1995).

Two basic processes are involved in AMD generation including abiotic chemical oxidation and microbiologically mediated oxidation of sulfide minerals (Lowson, 1982; McKibbon *et al*, 1986; Nordstrom, 1982; Singer *et al*, 1970; Snoeyink *et al*, 1980; Williamson *et al*, 1994). These are thought to include several types of oxidation-reduction reactions, hydrolysis, complex ion formation, and solubility controls (Nordstrom, 1982). Most researchers continue to recognize the basic system of biologically mediated chemical reactions as one that it is extremely complex, both thermodynamically and from the standpoint of kinetics (Evangelou, 1995; Williamson *et al*, 1994). This is more true in AMD's expression in natural environs where depositional environment, paleoclimate, modern climate, geology, hydrology, and man vastly complicate AMD expression (Caruccio *et al*, 1974; Cecil *et al*, 1985; Curtis *et al*, 1973, 1979; Dickens *et al*, 1986, 1989; Leopold *et al*, 1964; Nordstrom, 1982; Schumm, 1977; Skousen, 1996).

2.18.1 Processes of AMD Generation

A key feature AMD, in the absence of oxygen (depleted oxygen) or in the presence of oxygen, once initiated, the by-products of AMD reactions (e.g. generation of Fe^{3+}) can subsequently chemically oxidize pyrite. This generates more Fe^{2+}, generating more Fe^{3+} again, and again and again, until the pyrite is finally consumed through oxidative processes. The long-term discharge of acid leachates from AML spoils and excavation zones are therefore major environmental problems associated with disturbance of the soil and bedrock horizons (Daniels *et al*, 1995; Stewart *et al*, 1992). Once pyrite is exposed to O_2, the process is initiated, and eventually oxidation of pyrite will continue to (near) completion (Daniels *et al*, 1995; Lowson, 1982; Nordstrom, 1982; Snoeyink *et al*, 1980). Stream acidification is likely to occur if surface and ground water are allowed to discharge from AML sites in areas without sufficient carbonate buffering control (USEPA, 1980; USDA, 1985, 1998).

As a common example, in the eastern United States, AMD is generated via exposure of pyrite through underground and surface mining conducted in and around coal seams (Pennsylvanian age deposits) (Nordstrom, 1982; Rimstidt, 1997; Williamson *et al*, 1994; USEPA, 1976a, 1976b). Reactions of AMD due to oxidation of pyrite involve two steps of acid generation (as oxidation of Fe^{2+} and as precipitation of Fe^{3+}). The two acid generation stages are oxidation of FeS_2 using either Fe^{3+} or O_2, and then precipitation of ferric oxyhydroxides as silt and oxide coatings instream. Pyretic sulfur accounts for 50% to 70% of total mineral sulfur, occurs as microcrystalline framboids, and is the primary agent in the generation of AMD (Caruccio, 1975; Rimstidt, 1997; Skousen, 1996; Younger, 1996). Once set in motion, AMD generation steps can still continue, either biotically or abiotically for a great deal of time under a variety of site conditions. It is conceivable this can persist for perhaps centuries if not longer, making AMD a difficult issue to control (Daniels *et al*, 1995, Nordstrom, 1982; Snoeyink *et al*, 1980; Williamson *et al*, 1994; Younger, 1996).

2.18.2 Controlling AMD Generation

Use of wet/dry inorganic/organic covers, neutralization, and sterilization/inhibition of *Thiobacillus ferrooxidans* can be used to control impacts of AMD (USDA, 1998). As an example of inhibition, slow release of anionic detergents have also been suggested as an inexpensive way to reduce pyrite oxidation and acidification (Kleinmann *et al*, 1981; Younger, 1996). In the British coal fields it was reported that currently pumping to dewater certain fields is cheaper than allowing AMD to ensue and dealing with treatment costs. In other smaller fields, surface sealing may be feasible to some degree (Younger, 1996). Meek on the other hand found that prevention for strip mining techniques led to higher unit treatment costs (Meek, 1996).

One way to summarize the interaction of biology and AMD mechanisms is to note that:

1. Acid generation rates are limited by microbes where the rates are insensitive to pH changes below 6 s.u. (Rimstidt, 1997; USEPA, 1980; Williamson *et al*, 1994), and

2. The ecological impacts of AMD are increased by decreases in pH below 6.0 (Allan, 1995; Gallagher *et al*, 1999; USEPA, 1980).

It is known that adverse impacts from AMD upon receiving streams are expensive to resolve completely, and are often prohibitive in cost to fully resolve (Daniels *et al*, 1995; Younger *et al*, 1996). AMD is really more a problem to control then a problem to eradicate (Daniels *et al*, 1995; Snoeyink *et al*, 1980; Younger, 1996).

2.18.3 Chemical/Biochemical Reactions of AMD

Summarized by Evangelou (1995), three interdependent approaches can be used to study pyrite oxidation: a) inorganic/biochemical reactions, b) electrochemical reactions, and c) reaction kinetics. None are sufficient in and of themselves. Two constructs can be taken to conceptualize and describe underlying reactions and rates that govern sulfide oxidation and acid mine drainage. One is considered a classical overview interpretation of pyretic acid mine drainage generation (Nordstrom, 1982; Snoeyink *et al*, 1980). This stoichiometric approach though does not address surface chemistry phenomena. Differences among researchers can be found regarding the importance of relative roles of indirect and direct metabolic oxidation and the interrelated role of surface chemistry (Evangelou, 1995). More recently it has been suggested that the interaction of dissolved oxidants and the pyrite surface is not actually site-specific, where anodic and cathodic reactions may occur at different locations on the mineral surface (Williamson *et al*, 1994).

Most researchers continue to recognize an underlying system of chemical and biological reactions as extremely complex, both thermodynamically and from the standpoint of kinetics (Nordstrom, 1982; USEPA, 1980; Williamson *et al*, 1994). This is particularly more true in AMD's actual expression in the natural environment where climate, geology, hydrology, and man, vastly complicate the picture of AMD (Curtis *et al*, 1973, 1979; Dickens *et al*, 1986, 1989; Leopold *et al*, 1964; Nordstrom, 1982; Schumm, 1977).

Two approaches that can be taken to view and describe the underlying reactions that govern sulfide oxidation and acid mine drainage are briefly summarized here. These are represented by the approaches given by:

1. ***Classic stoichiometrically balanced AMD equations using iron, and***
2. ***A series of more elementary cathodic and anodic reaction steps.***

The first approach is considered a classical interpretation of pyretic acid mine drainage generation (Nordstrom, 1982; Snoeyink *et al*, 1980). The second is an interpretation offered by Rimstidt (Rimstidt, 1997; Williamson *et al*, 1994) that attempts to reconcile additional observational data collected to date in this area. While potentially inaccurate in certain finer details not covered, the cathodic reaction pathway given below has been shown to be the rate-limiting step in AMD generation (Rimstidt, 1997; Williamson *et al*, 1994). As Rimstidt indicated, the transfer of electrons from the mineral (FeS_2) to the oxidant (O_2), at a cathodic site on the mineral surface is the rate limiting step.

The idea that the oxidant (O_2) is directly involved in the activated complex is generally supported by demonstrated experiments. The first is that the pyrite oxidation rate depends on the concentration of Fe^{3+} (Williamson *et al*, 1994) and the second major finding is that the rate of oxidation is proportional to the concentrations of O_2 and H_2O_2 (McKibbon *et al*, 1986).

2.18.4 Classical AMD Reactions

The following set of reactions is presented in Table 2.34 describing all generally recognized (significant) acid mine drainage reactions, as controlled by *Thiobacillus ferrooxidans*. Various other sulfide oxidizing microbes would be expected to interact in multiple pathways, but perhaps in different ways. The kinetics of other microbes may not be as well understood as they are for *T. ferrooxidans*. This is due to *T. ferrooxidans'* dominance in the AMD cycle and its fundamental control in AMD generation.

Table 2.34 - Balanced Acid Mine Drainage Reactions (Rimstidt, 1997)

Mineral Name(s)	**Reaction**
Pyrite-ferrihydrite	$FeS_2 + 3.75\ O_2 + 3.5\ H_2O \rightarrow Fe(OH)_3 + 4H^+ + 2SO_4^{2-}$
Pyrite-O_2	$Fe\ S_2 + 3.5\ O_2 + H_2O \rightarrow Fe^{2+} + 2H^+ + 2SO_4^{2-}$
Fe^{2+}-O_2	$Fe^{2+} + 0.25\ O_2 + H^+ \rightarrow Fe^{3+} + 0.5H_2O$
Pyrite-Fe^{3+}	$Fe\ S_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 16H^+ + 2SO_4^{2-}$
Melanterite precipitation	$Fe^{2+} + SO_4^{2-} + 7H_2O \rightarrow FeSO_4 \bullet 7H_2O$
Melanterite –szomolnokite	$Fe^{2+} + 7H_2O \rightarrow FeSO_4 \bullet H_2O + 6H_2O$
Melanterite –copiapite	$5FeSO_4 \bullet 7H_2O + O_2 + 2H^+ + SO_4^{2-} \rightarrow$ $Fe^{II}\ Fe^{III}_4\ (SO_4)_6(OH)_2 \bullet 20H_2O + 15H_2O$
Fe^{3+} - ferrihydrite	$Fe^{3+} + 3H_2O \rightarrow Fe(OH)_3 + 3H^+$
Ferrihydrite – geothite	$Fe(OH)_3 \rightarrow FeOOH + H_2O$
Geothite – hematite	$FeOOH \rightarrow 0.5Fe_2O_3 + 0.5H_2O$

2.18.4.1 Classical Stoichiometric Description of AMD

The classical description of chemical/biochemical reactions related to pyrite oxidation is highlighted in Figure 2.9 and 2.10.

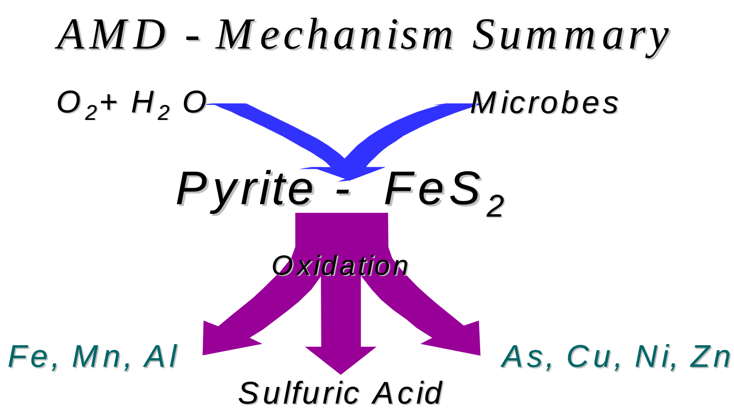


Figure 2.9 - Simplified Acid Mine Drainage Paradigm

Oxidation of Pyrite and Formation of Precipitates:

(1) $FeS_{2\,(s)}$ <> $Fe^{2+} + SO_4^{2-}$

(2) Fe^{2+} <> Fe^{3+} & S^{2-} <> SO_4^{2-} ***

(3) $Fe^{3+} + FeS_{2\,(s)}$ <> $Fe^{2+} + SO_4^{2-}$

(4) Fe^{3+} <> $Fe(OH)_{3\,(s)}$

*** Abiotic Oxidation. Catalyzed by Ferrobacillus & Thiobacillus Ferrooxidans (Autotrophic bacteria - speed up reactions @ low O_2/pH). H_20 is thought to be the source of oxygen in sulfate/hydroxides, not O_2 (Rimstidt, 1997).

Figure 2.10 – Classic Reaction Sets for Complete Pyretic Acid Mine Drainage Formation

2.18.4.2 Discussion of Classical AMD Reactions

The following briefly discussion encapsulates the process of AMD generation depicted in Figures 2.9 and 2.10. The actual process involves both biotic and abiotic reactions (Nordstrom, 1982; Snoeyink *et al*, 1980). Fe^{2+} indicates the reduced oxidative state, and Fe^{3+} indicates the oxidized ferric ion. These ions may also be referred to as Fe(II) and Fe(III), respectively.

- Reaction #1 in Figure 2.10 represents abiotic/chemical oxidation of pyrite when exposed to O_2 and H_2O. The reaction produces sulfuric acid and releases metals.

- Reaction #2 in Figure 2.10 represents a slow paced abiotic reaction in sterile conditions, otherwise microbial oxidation of Fe^{2+} by ferrooxidans can be over 10^6 times faster (Rimstidt, 1997; Snoeyink *et al*, 1980; Williamson *et al*, 1994).

- Reaction #3 in is an important alternate pathway of pyrite weathering with F^{3+} as the oxidant. Fe^{3+} is able to degrade pyrite without much O_2 present, while replenishing the Fe^{2+} --> Fe^{3+} reaction cycle.

- Reaction #4 in Figure 2.10 represents ferric iron precipitating as ferric hydroxide.

- Generically, the sulfide mineral source can include sulfide compounds of Fe, Mn, Cu/Fe, and As/Fe, and Reaction #4 can include other oxyhydroxides, sulfate and chloride salts.

- All the pyrite reactions above generate acid except for $Fe^{2+} \rightarrow Fe^{3+}$.

- AMD problems in the environment are typically not in equilibrium, and can endure for generations.

- Summarizing classical mechanisms and complicating factors, AMD involves:

  - Mineral extraction or disturbance
  - Chemical oxidation
  - Microbial oxidation
  - Hydrolysis and precipitation of various metal salts
  - Adsorption to clay and organic carbon and other metal substitutions can occur,
  - Iron and manganese oxyhydroxides tend to coat and cover surfaces in stream beds (Bigham *et al*, 1996; Diz, 1997; Gray, 1996; Nordstrom, 1982; Robbins *et al*, 1997)
  - Reactions are likely controlled by several parameters including pH, pE, CEC, solution activities, and common ion effects (Snoeyink *et al*, 1980).

2.18.5 Cathodic and Anodic AMD Reactions

The generic class of acid mine drainage (AMD) problems referenced originally can also be generalized (Rimstidt, 1997; Williamson *et al*, 1994). This can be done via discussion made with respect to oxidation of the iron sulfide minerals pyrite (FeS_2) and pyrrhotite (Fe1-xS). Discussion of these minerals can illustrate the key geochemical ideas and principles that apply to most if not all sulfide minerals (Rimstidt, 1997; Williamson *et al*, 1994).

The role of cathodic and anodic reactions have been proposed as mechanisms controlling abiotic and biotic oxidation of pyrite (Rimstidt, 1997). A diagram can be constructed for pyrite showing various kinetic rate mechanisms/reactions with respect to pH. Where conditions would allow, the microbial mediated oxidation of ferrous iron to ferric iron is kinetically favored.

A more detailed surface chemistry approach given below assumes similar features of the classical approach but explains AMD mechanisms in a step by step set of reactions, on an electron basis, imparting geological significance, and re-interpreting observational data. As indicated by Rimstidt, (Rimstidt, 1997) the oxidation of sulfide (S^{2-}) to sulfate (SO_4^{2-}) shows a transfer of 8 electrons per sulfur atom and the oxidation of disulfide (S_2^{2-}) to sulfate (SO_4^{2-}) shows 7 electrons transferred per sulfur atom. According to Rimstidt, the main idea of examining the oxidation process electron by electron is given historical support by the understanding that elementary redox reactions almost always behave this way. Rimstidt pointed out that semiconductor properties of the minerals involved, and the electrochemical nature of the reactions complicates the process. The discussion provided here attempts to establish a consistent framework that explains the underlying step by step elementary reactions as sulfur is oxidized.

2.18.5.1 Pyrite Oxidation

To summarize the electron transfer method of Rimstidt, the following points are taken directly from work attributable to Rimstidt's proposed cathodic/anodic reaction sets for oxidation of pyrite oxidation in AMD environments (Rimstidt, 1997):

- ✓ In the reactions in Table 2.34 for Pyrite-O_2 and Fe^{2+}-O_2, the mineral oxidation process does not oxidize iron; Fe^{2+} remains the same; in the mineral and in solution.

- ✓ In the AMD mediated reaction of Fe^{2+} to $Fe^{3+}$by *Thiobacillus ferrooxidans*, only sulfur changes valence going to sulfate as a predominant species.

- ✓ Isotope studies have shown that the oxygen found in that sulfate originates from the water molecule, not O_2.

- ✓ Due to the free energy of the reactions in Table 2.34 for Pyrite-O_2 and Fe^{2+}-O_2, they are favored strongly to occur; the forward direction is the only one considered. This is supported by kinetic considerations that show sulfate reduction is kinetically not favored below 300 ^{o}C.

2.18.5.2 Oxidative Dissolution and Sulfur Enrichment

In looking at pyrrhotite, an additional complexity is revealed, common in sulfide minerals, where Fe^{2+} is ejected into solution (Rimstidt, 1997):

- ✓ The oxidative dissolution reaction of pyrrhotite (Fe1-xS) + $0.5O_2 + 2H^+ = FeS_2 + Fe^{2+}$ produces a mineral with higher sulfur content at the surface.

- ✓ The importance indicated is that the initial stages of sulfide oxidation usually always involve an enrichment of sulfur at the complex surface.

2.18.5.3 Cathodic - Reaction: The Rate Limiting Step

A main result of the collective evidence placed together over decades of work by scores of researchers, a rate limiting step in the oxidation of sulfide minerals was eventually recognized (Rimstidt, 1997; Williamson *et al*, 1994):

- ✓ The limiting step is a transfer of an electron from the oxidant to a cathodic site on the mineral surface of the Fe^{2+} complex.

- ✓ The idea that the oxidant (O_2) is directly involved in the activated complex is generally supported by experiments that demonstrated pyrite oxidation rate depends on the concentration of Fe^{3+} (Williamson *et al*, 1994), and the rate of oxidation is proportional to the concentrations of O_2 and H_2O_2 (McKibbon *et al*, 1986).

- ✓ The nature and location of the cathodic sites is not well understood yet, but is believed to mostly involve cations; the analogy for pyrite: the as yet not well-described activated complex involves the transfer of an electron to Fe^{3+} absorbed from solution from Fe^{2+} in the mineral surface.

- ✓ The Fe^{2+} is released again to solution and an electron moves from an anodic site to reduce Fe^{3+} back to Fe^{2+}. This back and forth keeps occurring where in the end four electrons are transferred, but leaving the cathodic site the same as when the reaction started.

2.18.5.4 Anodic Reaction – Sulfur Oxidation

The driving force of mineral oxidation is the oxidation of sulfur itself (Rimstidt, 1997; Snoeyink *et al*, 1980). The anodic reaction has been shown to be constrained by the following aspects (Rimstidt, 1997):

- ✓ The driving force of the AMD reaction is oxidation of sulfur (S^-) → (S^{6+}), transferring seven electrons from the reduced disulfide sulfur atom.

- ✓ The original cathodic reaction causes the terminal sulfur to become more electropositive.

- ✓ The anodic site activated complex reaction is described by the equation: complex—S-S → complex—S-S^+ + e^- .

- ✓ The negative ends of the water dipole then attack the now electropositive sulfur. This is referred to as a nucleophilic attack.

- ✓ As an electron is transferred into the oxidant from the cathodic site, a hydrogen ion is released to solution to balance the reaction when S-OH is formed.

- ✓ The next two reactions involve: complex—S-S^+ + H2O → complex—S-S-OH + H^+ → complex-S-SO + e^- + H^+.

- ✓ A complex series of similar reactions ensue with progressive oxidation of sulfur producing a final anodic reaction described by: complex—S-SO_3 → complex—S + HSO_4^- + e^- + H^+.

- ✓ Four (4) water molecules are consumed and eight (8) hydrogen ions are produced per sulfur atom oxidized from disulfide.

2.18.6 AMD Expression Instream

Drawing a wider picture, the ecological relevance of the reactions of AMD and its interaction with landscape ecology are discussed briefly. On overview of dominant consequences of AMD in aquatic systems is given in Figure 2.11. AMD progression includes dissolution of pyretic materials and formation of sulfuric acid, and oxidation/reduction, chemical neutralization, and precipitation of various iron, manganese, and aluminum compounds (Lowson, 1982; Nordstrom, 1982; Rimstidt, 1997; USEPA, 1976a, 1980; Williamson *et al*, 1994). Due to resource extraction activities, the normal kinetics of undisturbed pyrite is eventually overtaken by relatively rapid oxidation (Snoeyink *et al*, 1980; Williamson *et al*, 1994). Land disturbance and these ensuing processes impact both physical and chemical qualities of the stream system.

Coal and hard rock mining lead to accelerated transport of groundwater rich in acid, iron, sulfate, and other minerals away from the zone of disturbance. Acidification of surface streams receiving AMD will subsequently occur without sufficient carbonate buffering control (Daniels *et al*, 1995; Skousen, 1996; USEPA, 1980). The long-term discharge of acid and metals from AML spoils and excavation zones therefore represent major environmental problems of coal and hard rock mining. AMD discharges instream can persist over decades to millennia for a given disturbance, and are recognized today as difficult and often prohibitively expensive problems to control (Cherry *et al*, 1999; Daniels *et al*, 1995; Meek, 1996; Skousen, 1996; OSM, 1999; USDA, 1999; Younger, 1996).

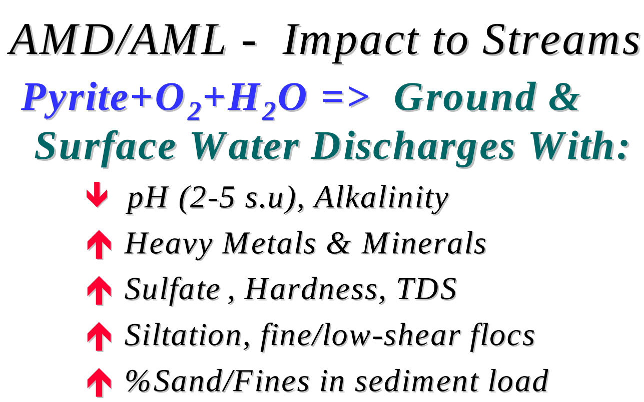


Figure 2.11 – AMD/AML Impact To Streams

2.18.6.1 Mechanisms Relating the Fate of AMD

Both solutes and flocculants generally disperse from the site via the stream as dissolved load, washload, suspended solids, or bed-load transport (Vanoni, 1977; Yang, 1996). Derived from AMD, low-shear iron and aluminum precipitates, representing very fine silt particle sizes, will normally transport as washload. Settled Al and Fe precipitates can be found in less turbulent niches of the stream channel during lower flows. Determined by sediment transport mechanics, degree of deposition of AMD precipitates will be strongly influenced by many hydrologic-hydraulic factors. These include proximity to abandoned mine land, pre-mining stream velocities, and other pre-mining channel geometry parameters (Bicknell *et al*, 1993; Higgins *et al*, 1987; Vanoni, 1977). Gradually downstream toxicity and physical impairment from sediment will decline through reduced unit-area discharge of AML sediment and AMD, defining a process of natural recovery of biota (Herricks *et al*, 1974).

Dilution of low pH and soluble metals associated with baseflow discharges during run-off follows. Abrupt transitions from low to high pH at confluence points of AMD impacted tributaries, or due to run-off, can result in decreased metal solubility and significant loading of oxidized metals to sediment (Snoeyink *et al*, 1980). These loads contain the final mineral deposition of pyrite as iron oxyhydroxide materials within sediment bars (Barret *et al*, 1993; Nordstrom, 1982; Yang, 1996). Hydrologic simulation and advanced modeling of these processes in general can be quite complex, where methodical approaches in determining partitioning dynamics must always be advocated (Bedient *et al*, 1992; Boult *et al*, 1984; Higgins *et al*, 1987; McKeon *et al*, 1988; Singh, 1995).

Qualitative scoring systems for coating of substrate surfaces by AMD based on physical sedimentation processes have been offered (Gray, 1996). Low pH, high specific conductivity, and gray, tan, yellow, orange, and red colored iron and aluminum mineral flocculants are also good indications of nearby AMD distress in local streams (Bigham *et al*, 1996; Evangelou, 1995; Nordstrom, 1982; Skousen, 1996). Typically, yellow/orange floc indicates lower pH ranging from 2.5 to 5 s.u.. In natural streams, floc color may not always accurately reflect a consistent pH though due to formation of various complexes. This can result due to variation induced by local geology and associated variable buffer control. Downstream sediments in Leading Creek eventually turn red-brown in color indicating the disposition of oxidized source pyretic iron as hematite coated quartz and feldspar materials. These oxyhydroxide-coated surfaces represent the final stage of instream precipitation of iron oxidized from pyrite for a given cycle. This cycle can re-occur instream through processes of de-acidification of anaerobic sediment columns, rendering neutral sediment pH when water column pH is depressed (Mills *et al*, 1989).

2.19 IMPACT OF AML-SEDIMENTATION AND AMD

A notable departure from a typical river study is large unreclaimed mining spoils occurring on steep to moderately sloping terrain. Optimal modeling of AML's adverse impacts upon Leading Creek is discussed below within the context of spatial scale and idealized fluvial elements. Surface contour mines (also referred to as strip mines) are also defined more commonly as abandoned mined lands (AML). AML may include abandoned underground and aboveground mining operations, where both may also be referred to generically as "resource extraction" operations (OEPA, 1991a, 1991b). According to the Meigs County soil survey, no known active surface mining is currently underway in Meigs County (Gillmore *et al*, 1991).

In historical studies of the Ohio River Basin and its watersheds, AML was often found to present the single greatest risk to aquatic ecology in any watershed. There were three primary effects of AML found and often noted from one watershed study to another study (OEPA, 1991a, 1991b; USDA, 1983, 1985):

- Systemic sand/fines deposition in and below AML tributaries,
- Local acidification, and
- Local water column metal toxicity.

Pointed out perhaps only subtly in this AML trilogy, dilution has to date been the apparent status quo treatment of AML associated AMD in the Leading Creek Watershed and most other watersheds in Ohio afflicted by AMD. Qualifying AML sand deposition, it can be associated with spoils from surface mines or waste materials brought to the surface by abandoned underground mining operations. Active SOCCO underground mine operations manage surface water run-off using at least 10-year storm basin detention design which should remove most suspended materials in compliance with permits associated with those operations. Coal and clastic dust particulate settled on foliage off-property may also be washed off during storm events suspended in run-off waters.

In a simple analogy, with decreasing energy slope the stream can efficiently only transport smaller and smaller particles downstream. Depending on how steep the local terrain is, AML sands from the original sandstone cap rock will eventually slow down somewhere in the channel, and eventually stop moving somewhere farther downstream (Schumm, 1977; Vanoni, 1977; Yang, 1996). Preliminary assessments of Leading Creek indicate an adverse impact of channel inundation by AML sands has occurred in many tributaries and the lower mainstem. Strikingly, substrate degradation is apparent to biota significantly farther downstream than is localized AMD partitioning in the water and sediment columns. AMD impacts can be diluted well upstream of regions impacted sand deposition (Leopold *et al*, 1964).

2.19.1 Regional Areas of Coal Mining and Impact

Summarized by Leading Creek researchers and work from other area projects on AML, the following points are highlighted to reflect the staggering reach of AMD in the Appalachian landscape (Cherry *et al*, 1995, 1999; OSM, 1999):

- Abandoned mined land discharges (AMD) draining from abandoned mined lands (AML) can produce devastating effects on the environment in the form of altered water chemistry and sedimentation. In some parts of Appalachia, acid mine drainage flowing from abandoned coal mines has caused pollution so severe that plant and animal life in many streams cannot survive (Cherry *et al*, 1999; OSM, 1999).

- The Environmental Protection Agency has singled out acid drainage from abandoned coal mines as the number one water quality problem in Appalachia (OSM, 1999).

- Many AML/AMD problems are the result of coal production that helped build America's industrial base and fueled our war efforts during World Wars I and II, many years ago (OSM, 1999).

- Acid drainage has had a devastating impact on people's lives and the vitality of the local economies. The benefits of cleaning up the rivers and streams can be clearly observed. One goal of the clean-up of acid drainage is the benefit for local economies arising from tourism and outdoor recreational activities such as hiking, camping, fishing, and boating brought back by clean streams (OSM, 1999; Smith *et al*, 1998; USDA, 1998).

- The eastern half of the United States includes three major bituminous coal regions, the Appalachian, the Eastern Interior, and the Western Interior. The Appalachian coal fields cover approximately 72,000 square miles in parts of nine states, including the eastern half of Ohio (USEPA, 1976a, 1976b; Vogel, 1981).

- The coals of Appalachia are of Pennsylvanian Age, and are essentially coextensive with the Appalachian Plateau physiographic province (Cecil *et al*, 1985; Wilson, 1998). The most abundant coal-bearing rock types in Appalachia are the fine-grained siltstones and shales (Vogel, 1981).

- Patterns of acid-mine drainage affecting major stream systems indicate that this problem is most prevalent in Pennsylvania, portions of eastern Ohio, a band along the boundary of Kentucky with West Virginia, Virginia and an area in north-central Tennessee (USEPA, 1980; Vogel 1981).

- Stabilization and reclamation of coal refuse disposal piles is an expensive dilemma still troubling the Appalachian coal industry today (Daniels *et al*, 1995).

- Surface mines, underground mines, mine tailings, and smelter waste dumps can discharge toxic materials and sediments that degrade water quality (Cherry *et al*, 1999; Daniels *et al*, 1995). AML and AMD impact upon the environment has led to various government agency programs designed to clean up abandoned mine sites.

- The mining industry is also actively seeking solutions to better deal with today's waste management problems. For instance microbes now are being ingeniously exploited to mine metal ores in piles (Ehrlich *et al*, 1990).

- Although stream pollution from acid mine drainage has been recognized as a major problem in the eastern United States for decades, the Appalachian Clean Streams Initiative of 1994 is the first coordinated effort with a primary focus of eliminating acid mine drainage. This initiative is a multi-agency effort involving federal, state, and local governments in cooperation with citizens, corporations, and universities to clean up acid mine drainage in Appalachia.

- ♦ According to OSM, the Appalachian Clean Stream Initiative's most challenging problem faced is acid mine drainage.

2.19.2 AML/AMD Best Management Practices (BMPs)

According to the Office of Surface Mining (OSM, 1999), the most common method used to eliminate acid drainage from abandoned underground mines is chemical treatment, which is expensive and requires constant maintenance. A second treatment method, still experimental but much less expensive is based on biological control. The newer technology, developed by the U.S. Bureau of Mines, diverts the flow of acid drainage through artificial or man-made wetlands. Biological processes remove excess acidity and iron from the acid drainage before it leaves the wetlands. Biological treatment is relatively inexpensive to construct and has been very successful on some small discharges of acid drainage (OSM, 1999). Another experimental method noted by OSM is the use of alkaline waste products from co-generation power plants to fill underground mines. Filling mine voids with this material is thought to eliminate production of acid drainage.

The purpose of mined land reconstruction is to stabilize the area and to restore it to a productive, economic use (Cherry *et al*, 1999; USDA, 1985, 1998). Objectives of reconstruction can cover six potentially overlapping viewpoints (USDA, 1998):

- Restoration of an economic land use,
- Restoration of desirable vegetation,
- Enhanced water quality and quantity,
- Improved aesthetics,
- Improved fish and wildlife habitat, and
- Improved safety and health for humans and other animals.

Factors needed to assess reclamation may cover specific physical and chemical properties describing the landscape and spoil piles (Daniels *et al*, 1995). USDA also indicates that factors limiting the use of specific techniques can include (USDA, 1998):

- The size of site to be reclaimed,
- Type of mining techniques used,
- Type of mine spoil on the site, and
- Water quality characteristics of the sites AMD.

Information from the Virginia Department of Mineland Reclamation and a study completed for the Ohio Department of Natural Resources were used by Leading Creek researchers to summarize available reclamation techniques that might be used for the Leading Creek Improvement Plan (Cherry *et al*, 1999; USDA, 1985, 1998). Summary guidance, technique by technique, is given next in the following excerpts from the Leading Creek Improvement Plan Study (Cherry *et al*, 1996, 1997, 1999). A summary and discussion then follows to bring to light critical AML-AMD reclamation design issues summarized from the work of Daniels and others.

2.19.2.1 Passive AMD Treatment

Passive treatment includes the use of an anoxic limestone drain (ALD) which is a buried bed of limestone gravel that generates alkalinity through the dissolution of limestone. The quantity of limestone included in the ALD is calculated from 25 years of expected limestone dissolution plus the targeted performance under design high flow conditions. Calcitic limestone with at least 85% $CaC0_3$ content is preferred. The limestone aggregate is placed in an excavated rectangular pit, covered with 4-6 mm of plastic, and buried with 2-3 feet of soil or spoil. Mine water enters one end of the limestone bed and is collected from the opposite end by a manifold system. The water level in the ALD is maintained at the top of the limestone layer through proper positioning of the effluent pipe.

2.19.2.2 Vertical Flow Ponds

A vertical flow pond (VFP) is a combination of limestone and organic substrate that decreases acidity and generates alkalinity. Water flows from the surface, downward through the substrate and limestone gravel, and into an underdrain system. The recommended VFP design contains 24 inches of surface water, overlying 12 inches of organic substrate, which overlies 24 inches of #3 limestone gravel. The organic substrate is amended with limestone aggregate (25% by volume) to increase its acid neutralization capability.

The limestone used in all cases is at least 85% $CaCO_3$, and is placed at the bottom of the limestone aggregate bed. The manifold connects to a solid pipe that passes through the berm and rises to an elevation consistent with the designed water level. An emergency spillway is placed 24 inches above the design water level and provides the capacity for water storage during high flow events and allows the passive development of additional head. The freeboard of the berms above the emergency spillway eventually is 24 inches. Inside slopes in the system are generally 2:1 while outside slopes are 3:1.

2.19.2.3 Sedimentation/Settling Pond

A sedimentation pond is intended to collect iron oxide precipitates. For the systems proposed, iron solids accumulate in the ponds at a rate of 1-2 inches per year. To accommodate this accumulation over the 25-year lifetime of the treatment system, the depth of the ponds is at least 5 feet. Alternatively, the ponds can be designed in a manner to facilitate the periodic removal of iron oxide solids. This approach may be favored if recovery and sale of the iron oxide is feasible.

2.19.2.4 Wetland

A wetland is intended to polish the discharge of a sedimentation pond or vertical flow pond. The wetland is constructed with a fertile substrate and planted with emergent wetland plant species (typically cattails and bullrushes). Water depth usually is 4-6 inches. The water level in the wetland is maintained by the effluent structure that can be gradually raised if the accumulation of organic matter and sludge causes short-circuiting of flow paths. Iron solids accumulate in the wetland at a rate of approximately 0.3-0.5 inch per year. Berms are sized to allow the accumulation of organic matter and iron sludge over the lifetime of the system. Historically these have only been viable for strong AMD seeps where low inflow exists.

2.19.2.5 Resoiling of Mine Spoil

For AML soils that are acidic, sandy, shaly or consist of coal waste, treatment with lime and/or fertilizer will not usually be sufficient to establish and maintain vegetation. When such conditions exist, resoiling is a potential method of providing an adequate root zone. Resoiling consists of the removal of topsoil from an uncontaminated site and spreading it 6 to 8 inches over the abandoned mined site. When the topsoil is in place it can be mixed with lime and fertilizer to promote vegetative growth (Daniels *et al*, 1995; USDA, 1985). Discussed further below, when reclaiming with less than standard 4 foot caps, some site specific knowledge of physical and chemical properties is usually needed to be successful (Daniels *et al*, 1995).

2.19.2.6 Preparation and Seeding of Existing Mine Spoil

Most abandoned coal mine lands have uneven surfaces which are gently sloping to very steep. Gullies ranging from 1 to 40 feet deep commonly occur on the mine spoil landscape contributing substantial amounts of sediment to streams and lakes. Grading is the first step in reclamation, followed by filling of gullies with adjacent soil until a maintained slope of 3:1 is obtained. Surface and/or subsurface drains are then installed if needed to control runoff. Surface drainage measures can include permanent diversions, straw bale diversions, grass and cement or rock-lined waterways. Once the mine spoil has been prepared, seeding can take place. If acid conditions occur within the soil then chemical processes are undertaken to encourage revegetation. These scenarios are similar to many typically taken to handle final land disposal requirements for other solid and hazardous wastes (Sharma, 1994; USDA, 1985).

2.19.2.7 Chemical Application and Seeding

Alternatives to resoiling for treatment of acidic soils can be accomplished instead with papermill sludge, municipal sludge, and fly ash. Use of these materials may not be feasible in the Leading Creek watershed due to the unavailability of these resources (USDA, 1985).

2.19.2.8 Sediment Traps

Sediment traps are designed to retain water long enough to allow suspended sediments to settle out and to prevent them from overflowing into receiving systems. The life of sediment traps is limited and they require periodic maintenance to continue to function at 100% capacity. Sediment traps used in AMD areas are similar to those used for the retention of sediments from agricultural erosion.

2.19.2.9 Channel Work and Channel Maintenance

Channel work and channel maintenance includes dredging and the removal of vegetation and debris from the stream system. Dredging is a hazardous technique for sedimentation management and provides only temporary relief from sedimentation (USDA, 1985).

2.19.2.10 Mine the Remaining Coal and Spoils

Due to the small size of some AMD sites and their sometimes sporadic spatial distribution across the landscape, re-mining of the remaining coal and any spoils in the area may not be feasible in most cases. Re-mining can be costly due to the large amount of earthmoving and excavation required (USDA, 1985). With apparent ample supply capacity at this time, increasing efficiencies being discovered in the coal extraction process also may make the unit price per ton continue to decrease in the future. This occurrence would tend to make more widely spaced, entangled resources increasingly less desirable for the time being.

2.19.2.11 Neutralizing or Impermeable Barriers

For small AMD sites, including tailing or gob piles, an impermeable barrier of clay or lime can be used. Neutralization of acid spoils is achieved by selectively placing them during the re-grading operations and coating the material with a layer of clay or lime. After the completion of reclamation, the natural movement of water causes the lime to infiltrate into the acid spoil, neutralizing the acidity before the leachate reaches a receiving system. The use of clay prevents water from infiltrating into spoil piles.

The surface sealing/barrier procedure retards the formation of acid and decreases the amount of leachate leaving the spoils pile. Covering a mined area with an impervious blanket of clay to prevent water infiltration into deep mines would also have a significant effect on erosion and sedimentation (USDA, 1985). Surface sealing for smaller areas has met some success in Britain's coal fields (Younger, 1996).

2.19.2.12 Seal Deep Mines

Deep abandoned mines are not a major concern in the Leading Creek watershed except with respect to dewatering events. Because the majority of abandoned mines are near-surface underground and above surface strip mines, more focus is placed there. Sealing will likely not arrest iron oxidation and AMD formation (Snoeyink *et al*, 1980) but can be used to slow the process down.

2.19.3 Updated AML Spoils Reclamation Theory

AML and AMD are expected to significantly increase risks to aquatic ecology, more so than any other human activities in Leading Creek (OEPA, 1991a, 1991b; USDA, 1983, 1985). Serving as the primary passive treatment mechanism in place, AMD impacts downstream of incident discharge zones below AML are attenuated by dilution mechanisms.

2.19.3.1 Revegetation, Alkalinity-Addition, and Soil Development

Where feasible, revegetation efforts need to stress self-sustaining progression and provide for key annual maintenance steps during early attempts to establish plant growth. Good soil development progression is critical to long-term success of the "topical only" treatment approach to resolving AMD. A topsoil layer thickness (e.g. vegetated zone thickness) with less than 1 ft.

of suitable earth materials can be utilized, and is likely the optimal course of action for spoil areas inside the Leading Creek Watershed. If thin topsoil layers are used for reclamation in AMD impacted areas, however, care must be taken to assure adequate nutrients are present and that desired consistency and durability of cover can be established (Daniels *et al*, 1995).

Vigorous, self-sustaining, soil-producing perennial vegetation is the goal of topical AMD treatment efforts (Daniels *et al*, 1995). Without pile-specific data and an individualized treatment approach developed for each area, AML researchers recommend up to 4 feet of cover materials (Daniels *et al*, 1995). This is a staggering concept for management of potentially 4500 acres of AML in Leading Creek. Given the dynamics of Leading Creek, economical address AMD across the watershed would require one to differentiate field-scale drainages contributing the most acid and fine-sediment (sands/fines) per unit area.

Without contributing alkalinity during reclamation, AMD problems will likely persist as will localized lethal impairment of aquatic organisms typically associated with AMD in the near field (Skousen, 1996; USEPA, 1980). For disturbed areas of AML where AMD impacts are significant, attempting to augment sites to retain the most favorable topsoil conditions and robust plant growth possible is likely to be the best management practice. This will deliver incidental benefits to any AMD impact in these areas, while addressing overall AML erosion and sediment control (Daniels *et al*, 1995; Stewart *et al*, 1992). Alluded to directly above, it should be well established that any given hillside region is contributing significant AMD prior to reclamation.

2.19.3.2 Key Aspects of AMD Remediation

The degree to which abandoned surface (ASML) and underground (AUML) mining each contribute to AMD in any reach in Leading Creek was unknown prior to this work. Without respect to direct AMD derived from underground works, erosion of coal mining refuse piles on the surface and bare or sparsely vegetated ASML (strip mine) footprints were thought to remain highest in concern regarding AMD and AML sediment. Minimalist revegetation such as tree planting efforts will tend to mitigate some sediment and erosion problems, however, only minimal benefits will be gained from these efforts in reducing significant impacts of AMD (Daniels *et al*, 1995; Snoeyink *et al*, 1980). To date, the degree that re-forestation and establishment of grasses aids in attenuation of AMD impact to ecology is not well defined on large study scales (Daniels *et al*, 1995). With respect to AMD, the most important properties of reclaimed soil and refuse materials appear to be (Daniels *et al*, 1995):

1. Topsoil water holding capacity,
2. Potential acidity of the final combined soil profile.

- *Also playing important roles are lack of soil cover and resulting elevated soil heat. Elevated soil temperatures can hamper soil development processes in an acidic environment.*

- *Water holding capacity is primarily determined by particle size distribution.*

- *It has been shown that bedrock profile buffering capacity is also a useful determination of potential AMD impact (USEPA, 1980). Bedrock classification in this case probably represents a rough approximation of certain soil properties recommended for investigation by researchers (Stewart et al, 1992).*

Findings from the Powell River Project (Daniels *et al*, 1995) are supported by historic areas of research that have touched upon the AML issue, from a perspective of ecology, toxicology, and engineering control of AML source problems in the environment (Cherry *et al*, 1995, 1999; Daniels *et al*, 1995; Skousen, 1996). Importantly, for soil materials, current soil pH is not necessarily a suitable measure alone in estimating potential acidity via acid base accounting. This is because of potential for remaining pyretic sulfur and naturally occurring limestone contents (Daniels *et al*, 1995; Skousen, 1996; Stewart *et al*, 1992; USEPA, 1980).

In summary of research to date, the main factors of concern in surface derived AMD design are cover thickness, lime addition, nutrient addition, and hardy plant selection. These aspects may also be applicable to AUML design concerns attempting to prevent or control acid generation and discharge as seeps. For ASML, adequate incorporation of these elements in basic design approaches are needed to establish a foothold for later perennial plant cover materials including grasses and legumes. Health of the later rising plant communities determines associated benefits of attenuation of AMD, and overall erosion control of surface spoils and ASML footprints. Acidity, low water holding capacities of sand, and poor nutrient composition are limitations that need to be typically addressed in reclamation design (Daniels *et al*, 1995).

2.19.3.3 AML Sediment-Derived Impacts

In most cases, barrier strip approaches will be more economical for initial sediment control only objectives where AMD will still to be addressed at some point. Initial efforts to control physical issues should still be implemented where economically practical to also have the best prognosis for aiding reduced acidification of the stream. In practice this implies cover applications as opposed to barrier approaches. Success of long-term AMD treatment concerns will likely be optimized through efforts facilitating natural system processes and functions, with an eye to sound soil construction and revegetation strategies.

AML impacts result in acidic landscapes where fewer organisms are capable of inhabiting the relatively hostile soil and water conditions (Nordstrom, 1982). With regard to plant communities, some plants are better than others at colonizing acidic soil (Daniels *et al*, 1995). Plant preference strategy may need to be exploited in design where possible, evaluating long-term colonization potential, net contribution of alkalinity, and reduction of AMD discharges (Daniels *et al*, 1995; USDA; 1998).

2.19.3.3.1 Channel Storage of AML Sands/Fines

A long-term view can be taken with respect to instream storage of AML sediment. For systems the size of Leading Creek, it is reasonable to expect that existing instream storage of sediments can be efficiently and adequately washed out by natural stream mechanics over a period of 10 to 25 years. Significant improvements to ecology would also be expected before the end of the first ten years. As opposed to instream dredging for example funding for such efforts could be better spent addressing other severe, non-funded needs ecology will likely face in the same time frame, for example, ameliorating AMD. Time frames here are crude estimates and can be improved upon with more optimal AML inventories and modeling of hillside, valley, and instream morphological elements. Natural alluvial channels have exhibited overt changes in sedimentation within similar time frames (Dunne *et al*, 1978; Leopold *et al*, 1964). For a given $1 investment in ecology, it is more likely that AMD will remain a serious problem well after instream storage has been addressed, rather than vice versa.

A serious drawback of dredging and channel manipulation is the inability to measure effects of simultaneous changes made to the landscape which will also be necessary if restoration is to succeed. This would apply to efforts geared strictly at sedimentation control or efforts that also address AMD. While it will remain appealing under most scenarios that prioritize human desires to control natures use of the floodplain, channel modifications such as dredging and riprap would not be expected to improve ecological structure and function in Leading Creek.

2.19.3.3.2 Addressing AML Sediment Impacts

Immediate revegetation to establish annual cover at all disturbed surface locations should be pursued with tree plantings, direct seeding and topsoil augmentation where piles with < 20% sands/fines should be augmented directly with top soil as needed for establishment of vegetation (Daniels *et al*, 1995). This will reduce sedimentation and begin to improve water retention and consumption of oxygen in the root zones. These methods will help primarily to reduce permeability of the soil profile and transpire oxygen and water away from pyrite. These actions also initiate good soil development. Successful replanting will need to address nutrient addition and may be limited by phosphorous, not nitrogen (Daniels *et al*, 1995; Stewart *et al*, 1992).

2.19.3.4 Looking at AMD Derived Impacts

A varying distinction between gob and spoil piles of coal refuse and naturally vegetated areas of overburden at peripheries of surface mine sites may be the case in USGS strip-mine designated areas in Leading Creek (ODNR Landsat5; Stewart *et al*, 1992; USGS 7.5-minute Quadrangle Series). Mine gob and spoil areas of Landsat5 may also represent residues of abandoned underground mining operations (AUML) or active SOCCO operations, both concentrated near mine entrances. Unless properly managed, any of these AML residual material areas can contribute to AMD generation in channels (Curtis, 1973, 1979; Dickens *et al*, 1985, 1989). The chemical and physical composition of waste materials is critically important to the final impacts exerted by AML (Daniels *et al*, 1995). Remaining pyrite in now exposed or seams or disturbed rock layers can also cause significant AMD.

There is a great deal of variability in coal refuse pile character in moving from definitions of gob to spoil to more generic overburden and subsequently for overburden areas with minimal cover, modest cover, and those areas reforested, or otherwise managed. It may also be that significant variation will be experienced within these designations. Heterogeneity of soil profiles, associated fluxes, and total acid potential will be encountered because of variable physical and chemical properties also affecting the temporal and spatial distribution of AMD

(Daniels *et al*, 1995; Stewart *et al*, 1992; Skousen, 1996). This will be true from subshed to subshed in Leading Creek, and can be influenced strongly by topographic relief. Leading Creek is a watershed spanning 388km^2 where, on a subshed basis, over half the watershed is affected by AML sediment or AMD impact, and over half by agriculture activities.

2.19.3.4.1 AMD and the Balance of Alkalinity

Widespread impact of AMD discharges in Leading Creek and their impacts on aquatic ecology are believed to be related most to areas disturbed by abandoned underground (AUML) and surface mine operations (ASML), together comprising AML. These operations took place upstream of Leading Creek's perennial and non-perennial drainages, as indicated by USGS and ODNR mapping. Discharged effluents from SOCCO's active deep long-wall mining operations are actively managed and treated (USEPA, 1976a; 40 CFR 434). In contrast, AML upland areas appear to still contribute significant AMD from oxidation of pyrite contained within the overburden, bedrock materials and other residues left at the surface, and exposed below ground.

Without neutralization of ASML/AUML associated acidic discharges, acidification will occur to varying degrees at all points downgradient of these abandoned sites. Untreated AMD discharges will also have comparably higher metals concentration and loading in the water column, depending on local geology (Nordstrom, 1982; USEPA, 1980). Shown in Figure 2.9, elevated metals in AMD will typically be observed for iron, manganese, aluminum, and several other heavy metals dissolved from parent materials in and around coal seams, such as zinc, nickel, and arsenic, among other heavy metals. All downgradient flow paths generically represent potential sources of soluble metals comprising AMD reaching the stream system.

2.19.3.4.2 Water, Oxygen, Microbes and Acid Generation

In AML impacted zones, where greater amounts of water now flow through the soil and bedrock, it is probable that significant acid production will follow once pyretic oxidation is initiated (Snoeyink *et al*, 1980). It does not appear feasible that this process will ever be shut off completely. Summarized previously, there are believed to be two general stages of acid generation (Nordstrom, 1982; Snoeyink *et al*, 1980). The first stage of acid generation occurs with the oxidation of ferrous iron or Fe^{2+} to ferric iron or Fe^{3+} (the numeral alone indicates a positive valance state). The second stage of acidification occurs as precipitation of Fe^{3+} as iron oxyhydroxides (Bigham *et al*, 1996; Broshears *et al*, 1996). The acid-preferring iron-loving microbes are adept at optimizing available resources with minimal oxygen present. This makes AMD kinetics difficult to control completely (Evangelou, 1995; Snoeyink *et al*, 1980).

Realizing oxidation will move forward to completion, the impacts of AMD are directly dependent upon the interim capacities offered by buffering in the soil profile, reduced water flow, and the ability to prevent flux of oxygen through the disturbed upper and lower soil layers (Skousen, 1996; Evangelou, 1995). The recharge or flux of water, acid, nutrients, and oxygen through AML soils feeds the deeper acidic infiltrating moisture. This moisture eventually transmits as AMD groundwater and discharges to the stream drainage network at baseflow rates. This progression of the hydrologic cycle determines acidification of the stream channels.

Once discharged to surface waters, total net acidity is determined by the inherent buffering capacity of receiving waters. Impact to aquatic ecology appears to be at least moderately correlated with pH (Allan, 1995; Gallagher *et al*, 1999; USEPA, 1980). Kinetics of iron reactions involved appear to be dominated in natural environments by ferrooxidans, primarily *Thiobacillus ferrooxidans* and *Ferrobacillus ferrooxidans* (Daniels *et al*, 1995; Williamson *et al*, 1994). Surface flows of sulfuric acid and metal laden AMD waters generated by ASML and AUML can result also from direct run-off, as depression storage release, groundwater seepage (e.g. springs) and lateral flow through piles (Bedient *et al*, 1992; Daniels *et al*, 1995; Fetter; 1993, 1994; Linsley *et al*, 1975).

2.19.3.5 Powell River Project Research Results

Daniels and others have conducted significant study in the coal fields of southwestern Virginia and eastern Tennessee draining to the Powell River. The researchers have noted that up to 4 times the amount of limestone determined via acid base accounting (ABA) test results may be needed for effective treatment of coal refuse piles (Daniels *et al*, 1995; Skousen, 1996). Other important points of the research of Daniels and others include the following notations (Daniels *et al*, 1995; Stewart *et al*, 1992):

- ❖ Refuse materials can be direct seeded or successfully reclaimed with reduced topsoil depths of 6 inches to 24 inches (standard = 4 feet) if and only if their physical and chemical properties are well understood.

- ❖ Nutrient addition is needed for good soil and root zone development.

- ❖ Soil moisture and temperature are critically important to establishment of plant communities.

- ❖ Oxidation rate of pyrite is high at the root zone, if oxidation in this zone can be reduced through increased transpiration, greater overall reduced acidity for a given timeframe would be expected.

- ❖ Soil compaction is a leading cause of revegetation program failures and should be avoided. Leaving soils as loose as possible enhances plant colonization and growth.

- ❖ While vigorous stands of annual cover crops on direct seeded coal refuse materials are readily established and often are observed, diverse self-sustaining stands of perennial grasses and legumes after multiple seasons are difficult to establish.

- ❖ The latter stage of revegetation resulting in perennial production and maturation of soil development processes is critical to establishment of the long-term success of reclamation.

- ❖ To address AMD, proper lime addition is needed in addition to stabilization and revegetation to reduce erosion and flow of oxygen and water. Lime or other forms of alkalinity need to be delivered *in situ* to attain effective neutralization of acidic material piles (via use of bulk blending techniques or other delivery). Without lime or other effective alkalinity addition, AMD discharges from net acid piles may be expected to persist for decades and longer periods.

- ❖ The sediment and erosion control programs established via revegetation will not typically address AMD water quality problems.

2.19.3.6 AMD Project Design Considerations

AMD project design considerations should include knowledge of baseline site condition and reclamation status. Designs should use local soils where feasible to attain improved topsoil and cover specifications. Based on available soils and neutralization materials available, cover design soil thickness and mixture for hillside slopes and slope toes should attempt to optimize variables relating to reduced soil permeability and infiltration. The end result is desired reduced acid discharge (at baseflow and from storm seepage pulses). Approaches should address surface durability, use, and ecological function associated with final design specifications selected. The latter aspects would be guided by natural vegetation schemes that can offer improved net buffering capacity of the soil, growth and self-sustained soil development potential. Finally, low cost of initial installation and maintenance is desirable (Daniels *et al*, 1995; USDA, 1998).

Where slope allows, severe-AMD control designs may consider augmenting run-off control via equalization and some direct neutralization of piles. Because of their appearance and concentrated distribution, gob and spoil piles make logical choices for any direct intervention that attempts to introduce large quantities of alkalinity upland. Up to 1% to 5% by weight of lime addition may be necessary for some coal waste materials (Daniels *et al*, 1995). Land application of low metal content alkaline fly ash, paper-mill sludge, or sewage sludge may also eventually be shown to be environmentally beneficial in these cases (Daniels *et al*, 1995). These treatment alternatives offer avenues of a potential points of revenue generation that can help offset costs of AMD treatment, while improving overall aquatic ecology.

Equalization at AMD sites in the Leading Creek Watershed may also incorporate the following aspects:

- ❖ Small wetland design (Daniels *et al*, 1995).
- ❖ Other man made detention/release structures for collection of concentrated baseflow and storm seepage (USDA, 1998).
- ❖ Consideration and control of overall carbonate equilibrium chemistry limiting kinetic control normally provided by ferrooxidans (bacteria). Temporal variation in relative rates of photoreduction and oxidation also appear to influence iron behavior (Broshears *et al*, 1996).
- ❖ In addition to constructed wetlands treatment schemes (Stark *et al*, 1994), active novel package plant treatment using ferrooxidans and attached growth on quartz sand may also prove feasible and economical for small seeps (Diz, 1997).

2.19.3.7 AML Impact and Treatment Cost Summary

The character of the final deposition of ASML aggregate residues at mined sites resembles thinly covered landfill units. Landfill configurations may primarily fit categories typified by the terms “upslope”, “canyon” (or valley), and “aboveground” (Sharma, 1994). In Leading Creek, these fills appear to have been constructed at the front-slopes and toes of moderate to steeply sloped hillsides. The fills were apparently constructed of rock and soil materials falling by gravity to the sides and down-slope of excavation operations. Large-scale use of explosives in removing sandstone cap rock in Leading Creek was a common technique.

Compared to undisturbed conditions, there is increased oxidation occurring within remaining overburden in the fills as well as upslope areas that include the disturbed midslope and crestslope profiles of hillsides. Hillsides in Leading Creek are now steeper in some areas, making revegetation efforts more difficult. The clastic and sand fill materials derived from Pinegrove (Pn-unreclaimed, Pu-reclaimed) series are more permeable in comparison to the undisturbed Upshur-Gilpin (Ug) soil complex; by up to 1 to 2 orders of magnitude (Gillmore *et al*, 1991; Lucht *et al*, 1985).

It is estimated that many of the actions currently assigned and undertaken by other stream improvement efforts in the Leading Creek Watershed may only be addressing maintenance concerns associated with the root problems identified above (e.g. digging out culverts, etc.). These reoccurring maintenance based projects were organized through local, state, and federal efforts and grassroots community organizing efforts. Without address of problems originating outside of the channel on the landscape itself, a cycle of maintenance will continue, as will the associated maintenance needs and costs.

In regard to the AML sedimentation problems observed in the Leading Creek Watershed, efforts directed at the source of the problem on land, as opposed to in-channel manifestation are likely to be more successful. In this sense addressing disease rather than only the symptoms of the disease (Diplas, 1997; USEPA, 1998; USDA, 1998). Attention to landscape today is a more consistently recognized framework for addressing the interconnectedness of the watershed, down through various landscapes, down to riparian zones, out into the channel, and down onto and into the beds of our streams and rivers (NRC, 1992; USEPA, 1996c; USDA, 1998).

In Leading Creek, ecological functions are lost to system-wide sedimentation and localized poor water quality impacted by AMD. If only 25% of Leading Creek AML lands were manipulated at an estimated reclamation cost of $35,000 to $50,000/acre, risk managers would need a total of $40 to $56 million to begin to restore basic ecological functions in this watershed. In this analysis, this would represent an average cost of roughly $5000 to $7500 per resident in the watershed. It is emphasized that this level of reclamation will likely not adequately address AMD, primarily addressing only long-term erosion and sedimentation. Remediation costs are staggering compared to available resources and the prognosis for success remains limited in reaches severely impacted by AMD today. Given that stream improvement resources in Leading Creek ($2 million^{+}) are currently far greater than other southeastern Ohio watersheds, overall prognosis is currently poor regarding attempts to substantially address water quality based deterioration. Increased understanding of AMD processes and mechanisms and optimization of AMD control is needed.

2.19.3.8 AML-Sediment Treatment Time Frame

Considering a restoration cycle of twenty-five (25) years, attempts undertaken in the future to better map AML sedimentation sources, and AMD discharge zones, could likely be more beneficial to ecology than selecting among non-prioritized sites without study. For example, this will be the case selecting among hillsides of each subwatershed reach monitored, unless all hillsides in all subsheds of all subwatersheds can be fully reclaimed which is unlikely.

Based on more informal analysis, a 5 to 10 year time period might be expected for significant attenuation of channel residuals to be observed once sand/fines run-off has sufficiently been arrested throughout the landscape. It may take a period of up to several decades to see residuals disappear all together (Leopold *et al*, 1964). This is consistent with the fact that it in Leading Creek, heavy sand deposits have been present in all parts of the mainstem channel for at least 15 years. It would appear that many upland areas of the system are still unreclaimed, which is consistent with soil survey descriptions for Pinegrove soil series units describing ASML lands (Gillmore *et al*, 1991). U.S. Department of Agriculture Soil Conservation Service (SCS) personnel working in Leading Creek experienced similar time frames in observing positive benefits instream from addressing severe AML sediment problems.

2.19.3.9 AML-AMD Treatment Time Frame

Resolve of AMD impacts will require combined approaches looking at reduced surface erosion, reduced soil permeability, and addition of alkalinity (Daniels *et al*, 1995). Without direct addition of alkalinity, mitigating factors may include simple attempts to reduce infiltration on hillsides, and exploiting dilution at groundwater discharge points. Unless saturated by water, oxidation of pyrite in natural settings will be mediated by ferrooxidans and other microbes, and will continue for at least many decades (Evangelou, 1995; Snoeyink *et al*, 1980; Younger, 1996).

The actual time frame for acid attenuation will be dictated by the acid potential of the incident bedrock, buffering potential of cover soil materials, ferrooxidans' kinetic rates, and the associated natural carbonate buffering system at work. Successful control of AMD will also depend upon the degree wetlands and other approaches like equalization and dilution can be used in treatment schemes as final polishing steps. Once AMD is generated it is practical to assume its discharge to the land surface drainage network is inevitable at some concentration. Ecology can benefit more from viewpoints that focus on concentration as opposed to load rate. It is most likely that flow regime will become a primary point of focus in design and control strategies.

Without complete flooding of the hillside-valley to dictate oxidation controlled by diffusion (e.g. a reservoir water cover), attempts at shutting down oxidation of the hillsides will likely meet severe limitations and therefore limited success (Evangelou, 1995; Snoeyink *et al*, 1980). A 4 ft. thick (or highly designed 1 ft. thick), limey soil cover, vegetated with plants that have robust capacities for meeting a wide range of moisture, pH, and soil temperature conditions appears may be the best long-term approach at source control of AMD (Daniels *et al*, 1995).

2.20 ECOLOGICAL RISK ASSESSMENT ELEMENTS AND PARADIGMS

2.20.1 Environmental Effects Upon Biotic and Water Resource Integrity

Described in the context of integrated ecological risk assessment there are five major environmental effects which distinguish protection and classification of biotic and water resource integrity. These five effects incorporate the domain of potential human influence, induced alteration, and ultimate degradation. The effects were given Karr (1986) (USEPA, 1996c):

1 Energy Source

- Type, amount, and particle size of organic material entering a stream from the riparian zone versus primary production in the stream
- Seasonal pattern of available energy

2 Water Quality

- Temperature
- Turbidity
- Dissolved oxygen
- Nutrients (N-nitrogen & P-phosphorous)
- Organic/inorganic natural/synthetic chemicals
- Heavy metals and toxic substances

3 Habitat Structure and Quality

- Substrate type and quantity
- Water depth and current velocity
- Spawning, nursery, and hiding places
- Diversity
- Pools
- Riffles
- Woody debris

4 Flow Regime

- Water volume
- Temporal distribution of floods and low flows
- Flow regulation

5 Biotic Interactions

- Competition
- Predation
- Disease
- Parasitism

Yoder and Rankin have significantly expanded upon elements of Karr's basic five-point description (Yoder *et al*, 1996). The importance of habitat structure has been historically and more recently emphasized, spanning influences of regional geology and geography to the distribution of fines in interstitial pore spaces of riffle bars (Diplas, 1994; OEPA, 1989; Osborne *et al*, 1991; Rankin, 1989; Richards *et al*, 1994b, 1996; USEPA, 1980, 1989, 1996c).

2.20.2 Sediment Triad

The Sediment Triad (Chapman *et al*, 1990, 1992) based on sediment chemistry, toxicity, and biodiversity forms the basis of many ecological risk management criteria used in selecting restoration projects for riverine systems (USEPA, 1996c, 1998). Biodiversity and sediment toxicity measures, augmented with water column chemistry and acute toxicity tests are often the primary quantitative metrics investigated where biodiversity is considered as the primary dependent variable under investigation (Burton, 1992, Powers *et al*, 1992; USEPA, 1992, 1996c, 1998).

Figure 2.12 presents a composite paradigm of the Sediment Triad approach expanded for multi-media assessment that was applied in the study of the Leading Creek Watershed (Gallagher *et al*, 1999). The paradigm can of course be expanded to include pathology and bioaccumulation for instance (e.g. tissue sampling etc.), where these amendments represent more complex investigations (Burton *et al*, 1992; Chapman *et al*, 1992).

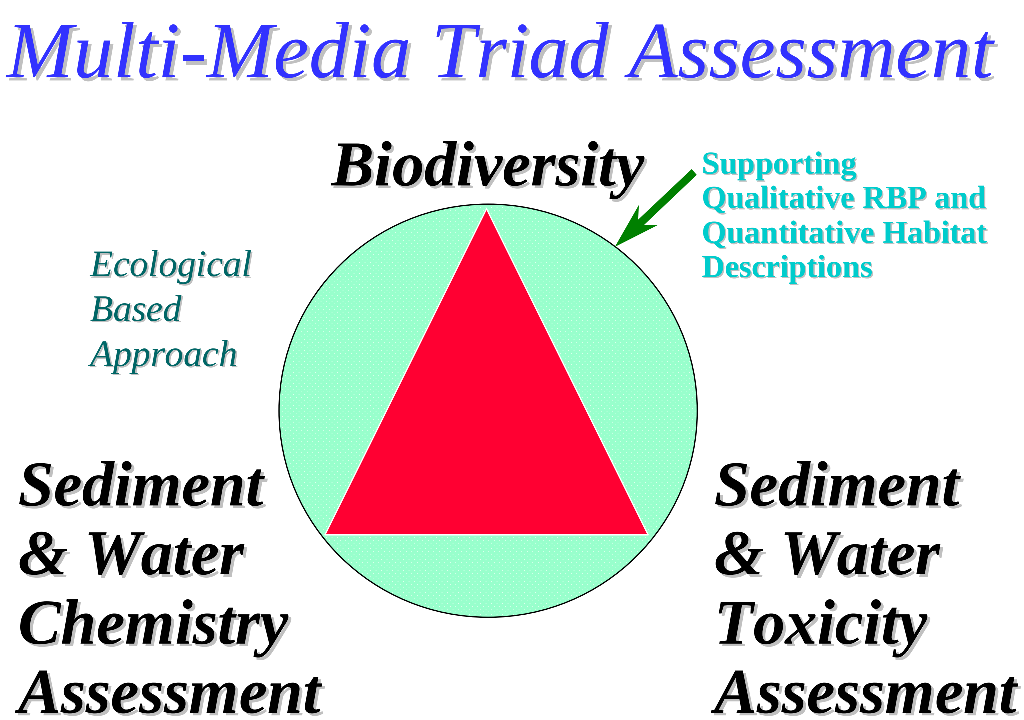


Figure 2.12 - Ecological Based Risk Assessment Paradigm
(Adapted from Chapman *et al*, 1990 1992; Gallagher *et al*, 1999)

Potentially directly utilized in the biodiversity-leg of the Sediment Triad are indexed lumped parameter scores such as IBI and ICI that represent biotic integrity. The IBI and ICI scores were introduced at the beginning of the discussion in Section 2.3 - Integrated Ecological Risk Assessments. In cases of IBI and ICI scores, it is interesting to note that quantitative metrics (like subscores in Tables 2.1 and 2.2; e.g. Total Taxa) may actually be needed to significantly correlate land use/cover patterns to changes in biodiversity or ecology (Richards *et al*, 1994b; USEPA, 1996c). Lumped integrative type scores like the IBI and ICI may account for more than can be simply explained in conventional land use/cover variables/functions, or other metrics of soil and water chemistry. The same may be true in predicting exceptional occurrences of biodiversity within a given ecoregion or subsystem.

Taxon richness is assumed to be inversely related to the degree of stress, whereas biotic indices attempt to summarize information on the tolerance of the macroinvertebrate community (Cherry *et al*, 1999; Lenat, 1993). The North Carolina (NC) Division of Environmental Management uses taxon richness of the most intolerant invertebrate groups (Ephemeroptera, Plecoptera, Trichoptera, or EPT) and a biotic index similar to that of Hilsenhoff's biotic index designed to detect the influence of oxygen demanding wastes on biota (Cherry *et al*, 1999; Hilsenhoff, 1987; NCDHNR, 1990; USEPA, 1996c). Similar biotic indexes created by others had also been created in earlier decades relating sedimentation impacts (USEPA, 1996c).

2.20.3 Trophic Relationships and Energy Balance

An overall trend of community structure may be expected to establish within eastern United States regions based on increasing stream order or drainage area (Minshall *et al*, 1985; Strahler, 1957; Vannote *et al*, 1980). The continuum concept has been slowly evolved as a model for describing the feeding ecology of running waters over the last two decades. USEPA points out that the original continuum description is not directly realized in most watersheds. Still, many of its underlying principles apply, particularly those regarding energy balance and aquatic ecosystem interaction (Cummins, 1983; Minshall *et al*, 1985; USEPA, 1996c; Vannote *et al*, 1980).

The biotic continuum represents a delineation of trophic structure of benthic invertebrates (Cummins, 1983; Minshall *et al*, 1985; USEPA, 1996c; Vannote *et al*, 1980). The continuum tracks a process of energy transformation along the stream. It was originally explained in part by a ratio of photosynthesis to respiration. A shift in this ratio occurs as stream order increases. In larger streams photosynthesis would be expected to dominate. Heterotrophic communities and respiration processes are expected to control in headwater areas, for instance converting leaf material down through the running water system making that energy available for invertebrates living within the higher order reaches (Allan, 1995; Cummins, 1983).

2.20.3.1 Heterotrophs

No food web exists for any stream but instead a number of partial food webs exist (Allan, 1995). Together various models can be constructed depending on actual forms of energy inputs but may include heterotrophic sources of coarse particulate organic matter (CPOM), fine organic particulate matter (FPOM), and dissolved organic matter (DOM).

For a woodland forest, inputs of organic matter to running waters would be described by the following elements, where autotrophic conversions to final heterotrophic pathways are marked by an asterisk (*) (i.e. photosynthetic input instream) (Allan, 1995):

- **Coarse Particulate Organic Matter (CPOM)**,
  - Leaves and needles
  - Macrophytes during die-back (*)
  - Woody debris
  - Other plant parts (flowers, fruit, pollen)
  - Other animal inputs (feces and carcasses).

- **Fine Organic Particulate Matter (FPOM),**
  - Breakdown of CPOM
  - Feces of small consumers (benthic macroinvertebrate)
  - From DOM by microbial uptake
  - From DOM by physical-chemical processes
  - Sloughing of algae (*)
  - Sloughing of organic layers
  - Forest floor litter and soil
  - Stream bank and channel.

- **Dissolved Organic Matter (DOM)**
  - Groundwater
  - Sub-surface or interflow
  - Surface flow
  - Leachate from detritus of terrestrial origin
  - Throughfall (e.g. canopy-precipitation interaction)
  - Extracellular release and leachate from algae (*)
  - Extracellular release and leachate from macrophytes (*).

2.20.3.2 Autotrophs

The counterpart to heterotrophic energy transformation is represented by the various autotrophic organisms that convert sunlight and non-living matter into energy. In the loctic energy system for running waters, important autotrophs are green plants and some bacteria. These include macrophytes or large plants, and smaller autotrophs classified into two groups: periphyton (found on substrates) or phytoplankton (floating in suspension). Periphyton is comprised of filamentous, gelatinous, prostrate, and crustose organisms. The last three organism forms are classified as vulnerable to higher trophic structural attributes of rasping and scraping. Filamentous organisms on the other hand are broken down into two groups defined by their vulnerability to higher trophic structural attributes of scraping-gathering and gathering-shredding-piercing (Allan, 1995).

The autotrophic and heterotrophic sources and ultimate transformation of energy into the stream system of running waters as described provide the basic feeding materials for higher trophic organisms. Consumer feeding roles can be used to describe the primary classification of various invertebrate consumers (Allan, 1995; USEPA, 1996c). These roles are commonly broken down along the following lines:

- Shredder
- Shredder / gouger
- Suspension feeder / filterer-collector
- Deposit feeder / collector-gatherer
- Grazer
- Predators (use animal prey as food resource)

The structural attributes given represent trait organizational aspects of benthic macroinvertebrates sampled via instream biosurvey techniques (OEPA, 1989; USEPA, 1989, 1996c). Total Taxa is one metric that spans across all of these groupings as a characterization of biodiversity. Another popular example is EPT scoring which represents various properties of sensitive insect families of Ephemeroptera, Plecoptera, and Trichoptera (e.g. EPT Taxa).

2.20.4 Ecotoxicological Endpoints

Ecotoxicological endpoints are intended to reflect ecosystem responses to sediment chemistry or other harmful stress. However, ecotoxicological endpoints often are insufficient by themselves in completing adequate risk assessment. Two major limitations exist in the Sediment Triad. First, relationships to stressor sources and related impact are not usually well defined nor quantitatively identified by the metric set, i.e. the source of the ecological stress is not identified, only its existence. Second, supporting physically based habitat measures are generally inadequate in describing most variation observed in biodiversity not attributed to toxin stress (Gallagher *et al*, 1999). Because of these aspects, single-test risk assessment strategies (Chapman, 1995; Long et al, 1996) can overlook significant system dynamics, though a single test strategy remains by itself a useful screening tool (Burton, 1992; Chapman, 1995).

2.20.5 Macroinvertebrate Surveys and Toxicity Testing

Macroinvertebrate surveys provide an indication of aquatic life living in the sediment as determined by field sampling procedures (OEPA, 1989; USEPA, 1989). Chemical contamination is most often found in fine “muddy” sediments, which characterize depositional areas (Powers *et al*, 1992). Examples of depositional areas in natural alluvial streams are pools, transition zones from pool areas, and point, tributary, middle, and alternating bars which exhibit lower energy and turbulence (Yang, 1996). Because benthic macroinvertebrates are good indicators of toxin impact to the overall ecosystem (Rosenberg *et al*, 1993), they are increasingly being used to assess sediment toxicity (Allan, 1995; Ingersoll *et al*, 1990; USEPA, 1996c).

Exposing borrowing organisms to contaminated sediments is thought to represent the worst case real world condition, though debate continues over the most appropriate uptake path to monitor (i.e. pore water, sediment, or water column/pore water interfaces) (Powers *et al*, 1992). All three phases are related and are critical to proper ecosystem risk assessment.

2.20.6 Sediment Toxicity Testing

Sediment toxicity tests may be influenced by sedimentation and grain size effects (DeWitt *et al*, 1988; ASTM, 1990a, 1995). For most macroinvertebrates, domination of bed sediments by particles <2 mm becomes problematic to species survival, and an alternate environment is sought via drift if transportation is available (Allan, 1995; Richards *et al*, 1994a; Vannote *et al*, 1980; Waters, 1995). Potentially perceived as toxicity, displacement effects can greatly effect bioassemblage measured at a given site. *Chironomus tentans* prefers fine sediment, which is also the particle size that tends to displace large numbers of other macroinvertebrate families in streams (Richards *et al*, 1994a, 1994b).

In areas with abandoned mined lands, C. *tentans* in-lab test responses might increase at sites with poorer diversity, showing possible tolerance for substrates, toxicants, and or acidity (USEPA, 1980). This would be consistent with USEPA's finding for acidified streams due to acid rain. C. *tentans* may also gain advantage found in sandier AML materials (Belanger *et al*, 1985), along with fine silt flocs available at AMD sites (Bigham *et al*, 1996; Gray, 1996), and silts and clays eroded from farmland and banks (Feusner *et al*, 1989; Gilmore *et al*, 1991; Lucht *et al*, 1985). Thus the midge's use as an ecological indicator could be potentially masked in some AML areas, or those areas commingled with agriculture and AML. C. *tentans* results might inadvertently dilute the perceived impacts of AMD and sedimentation in the risk assessment.

Problems with inconsistency in sediment toxicity test data were realized for some AMD influenced sites that did not have 100% mortality but were expected to when compared with *in situ* toxicity test results and benthic macroinvertebrate data (Babendreier *et al*, 1997; Cherry *et al*, 1995, 1999). The more recent sediment toxicity testing protocols by USEPA and ASTM are believed to reflect more environmental realism (ASTM, 1995; Cherry *et al*, 1997, 1998; USEPA, 1994). A pivotal assumption in applying the sediment triad is that each of the three measures in the triad are accurate and are representative of the site. If biases or other confounding factors invalidate the sediment toxicity test for example, then conclusions from the sediment triad are less strongly held. Efforts are currently underway to simplify toxin impact assessment through single-component analyses, i.e. use of the sediment toxicity portion of the triad (Chapman, 1995; Long *et al*, 1996; USEPA, 1992, 1995, 1998). Reliable and accurate chronic toxicity tests are essential to meeting these goals. There is more work to be done in the area of toxicity test development (Babendreier *et al*, 1997; Burton *et al*, 1992) and more continued verification of the promising state of the art *in situ* testing regimes (Cherry *et al*, 1996).

2.20.7 *In Situ* Clam Toxicity Testing

Asian clams have become ubiquitous to many riverine habitats in the United States. The Asian clam species has been show to be sensitive to environmental indicators of heavy metal stress including pH and temperature stress (Belanger *et al*, 1985, 1986, 1987, 1990; Cairns *et al*, 1983; Cherry et al, 1993; Cherry, 1996; Doherty, 1987; Farris *et al*, 1988, 1989; Graney *et al*, 1980, 1983, 1984; Rodgers *et al*, 1980). Asian clams have been used in transplant studies over the past two decades (Cherry *et al*, 1979a; Cherry, 1996; Farris *et al*, 1994). *C. fluminea* scavenge sediments, are able to travel at rates up to 250 cm/hr, and prefer a coarse sand material for burrowing (Belanger et al, 1985).

Asian clams can grow up to 4 mm per month as juveniles which is an ample amount of growth increment to indicate sensitive differences between recovered and stressed sites (Cherry, 1996). The tests appear to be sensitive to chemical impairment and may also have some influence from physical parameters such as particle size (Belanger *et al*, 1985) and more expectedly temperature (McMahon *et al*, 1986). Research of Asian clam populations has noted the basic spread of Asian clams from two distinct epicenters of artificial introduction into the waterways of the United States. The time frame for introduction was based on published reports of new populations from 1924 to 1982. The release areas occurred in the northwest and a second population was eventually introduced along the Ohio River (McMahon, 1983).

2.20.7.1 Existing *In Situ* Clam Growth Model

In development of the conceptual model currently used to describe Asian clam growth, researchers to date have been investigated several key issues. The following summary notes are given by (Bryne *et al*, 1989; McMahon *et al*, 1986):

- Investigators generally agree that *C. fluminea* may have two distinct reproductive cycles annually, where juveniles are released. This previously confounded growth rate and life span studies.

- A negative linear relationship between shell length growth rate and size may be expected. A positive exponential relationship between growth rate and temperature will be expected.

- Based on works by McMahon and others, it has been found typically that the growth of caged specimens is the same as that of marked free-living individuals. Cage materials were constructed of 5mm galvanized aluminum hardware cloth (mesh) (McMahon *et al*, 1986). This assumption was further successfully tested comparing caged results against results for the natural population.

- Many studies exist which show that for biota and clams specifically, modest rises in temperature stimulate growth.

- A later lab investigation looked into the exposure tolerance, aerial respiratory behaviors, and the rates of water loss of *C. fluminea*. These traits were evaluated against three temperature levels and five humidity levels to assess variation. Usually consistent but some mixed results were obtained, suggesting potential adaptive behaviors to certain chronic environmental stresses was possible in *C. fluminea*.

- Large inter-annual variations in growth rate can be due to availability in phytoplankton.

- Data from studies on the Kanawha River, West Virginia, near Leading Creek, show that growth rate and life span population in the eastern United States are well within ranges recorded for this species in other geographical areas of its range (Joy, 1985; Welch *et al*, 1984).

- ♦ Life span is thought to be approximately 3 to 4 years and possibly up to 6 years. Growth rates projected by McMahon can range 0.01 to 0.15 mm/day.

2.20.7.2 *In Situ* Clam Growth Models

A great deal of variability exists in analysis from study to study that might be explained by drainage area, which to date has not been accounted for by McMahon and other researchers (McMahon *et al*, 1986). It was noted earlier that a trend of community structure will establish within eastern United States regions based on increasing stream order or drainage area (Cummins, 1983; Dunne *et al*, 1978; Minshall *et al*, 1985; Strahler, 1957; USEPA, 1996c; Vannote *et al*, 1980). Works by Cummins, Minshall, Vannote, and others represent supporting evidence in hypothesizing that Asian clam growth may also be dependent in some way on drainage area.

Asian clams are uniquely established biotic metrics of ecotoxicology and can be quite revealing in terms of the nature and extent of system stress. It will be beneficial where possible to establish an accurate baseline reference for expected clam growth, both in the eastern U.S. and elsewhere. The Asian clam's potential viability *in situ* in streams across the U.S. presents the clam as an ideal candidate for environmental monitoring programs. The Asian clam's knack for showing similar growth traits throughout that domain may also make widespread growth model calibration simpler. For widespread application in ecotoxicological risk assessments, it would primarily be limited by extended temperatures above 30 ^{o}C (Mattice *et al*, 1985).

Three factors (age, temperature, and drainage area) have been suggested here to dominate growth rates over any standard duration test. The existing conceptual model for clam growth represents growth as a function of only age and water temperature (McMahon *et al*, 1986). A fundamental disparity also exists between temperature impact assessment where McMahon's model is exponential over all ranges, and Mattice suggested that decreases in growth occur above 25 ^{o}C (McMahon *et al*, 1986; Mattice *et al*, 1985). Overall, McMahon's approach conceptualizing normal growth rates without toxicant impact has been determined from development of a simple model of individual growth rate in *C. fluminea* based on regression against shell length and ambient water temperature (McMahon *et al*, 1986). A family of curves was developed for individuals at 5mm, 10mm, 20mm, 30mm, and 40mm shell lengths, and in a second approach for water temperatures of 5 ^{o}C, 10 ^{o}C, 15 ^{o}C, 20 ^{o}C, 25 ^{o}C, and 30 ^{o}C.

In the presence of water column toxicants, the novelty of the *in situ* clam test is that the clams will reduce siphoning intake, in essence clam-up, and fail to grow at expected rates or perhaps die (Cherry, 1996). A problem though can be variability of results at sites with seemingly similar, good water quality. Low growth and exceptional growth are not always successful indicators of relative concern (Cherry *et al*, 1995, 1996, 1997, 1999).

Based on initial *in situ* clam growth results found in Leading Creek and shown in Figure 2.13, evidence suggested that *C. fluminea* followed a predictable but unaccounted for trend in expected clam growth for a given age and temperature. The rise in growth moving downstream indicated a quantitative relationship with catchment drainage area. The data represents the first 35-day test conducted in Leading Creek in June of 1996, where water flowrates were relatively strong. Difficulty in explaining clam growth in this example was the implication in Figure 2.13 that a significant pollutant source existed at the headwaters and was diluted going downstream. This is an unlikely scenario compared to the potential that *in situ* clam growth rates were dependent on some energy continuum (Karr *et al*, 1986; USEPA, 1996c).

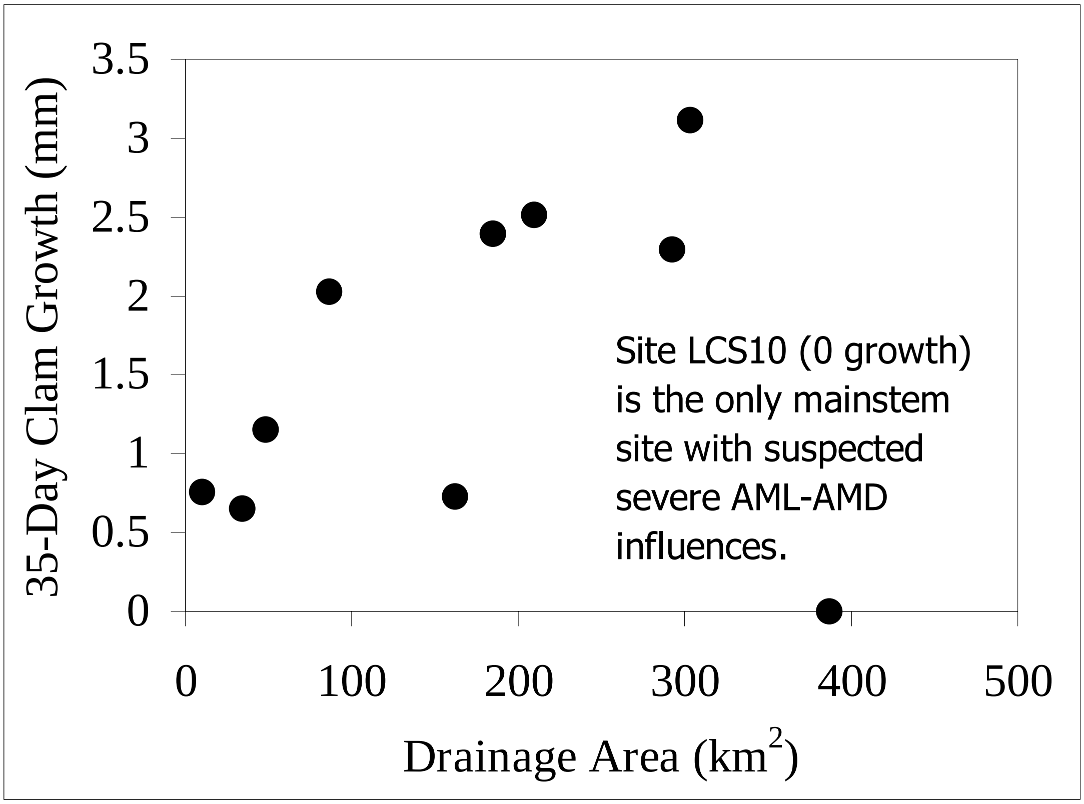


Figure 2.13 – Leading Creek Mainstem *C. fluminea* 35-Day *In Situ* Growth - June 1996

The dominant trend noted in Figures 2.13 might follow flow quantity or variation, which also would be expected to be related to drainage area (Dunne *et al*, 1978; Fetter *et al*, 1994; Leopold *et al*, 1964; Yang, 1996). Several tests conducted under various flow regimes could better evaluate the relationship between flow rate, drainage area and growth rate. Velocity might also be a parameter under suspicion based on how clams are placed within a given cross-section. Clams are typically placed in stream looking for conditions that will support adequate flow-by and water depth, defining appropriate habitat conditions throughout the test.

2.20.7.3 Need For an Unbiased *In Situ* Clam Growth Model

The novel value of correcting *in situ* clam test results for spatial dependence would be to develop an unbiased estimator of chemical (negative) or nutrient (positive) impact on normal growth rates. Thus, it is conceivable that abnormal results can identify sites with toxic loading and nutrient inputs. McMahon (McMahon *et al*, 1986) studied *C. fluminea* from Texas, where other researchers have studied the clams from eastern and western states. In McMahon's work, correction curves for temperature were provided across cold 5 ^{o}C to warm 30 ^{o}C temperatures for various age clams. These representations may be inaccurate where they do not appear to correct for or account for variation in drainage area, and where temperature itself may be in adequately reflected in exponential curves for temperatures above 25 ^oC.

The spatial dependence concept introduced here for Asian clams in loctic systems is consistent with that prescribed by USEPA for multimetric approaches for biocriteria development (USEPA, 1996c). In those recommendations, evaluation of scale effect is conducted through graphical analysis of biometrics plotted against a chosen continuous covariate such as stream order or drainage area. Noted earlier in discussion of biotic integrity, such scale corrections for biometrics are often necessary (Karr *et al*, 1986; 1991; OEPA, 1989; USEPA, 1989). Scale dependence issues are all related to the basic continuum concept put forth to date (Vannote *et al*, 1980). As McMahon pointed out, there is also a general need to establish population differences, if any, defined for instance by the line drawn by the Mississippi River. Genetic differences may have some influence on the growth rates and life span (McMahon *et al*, 1986).

2.20.8 Habitat Evaluations

Because ecological stress is not necessarily derived solely from toxin impact, current ecological research recognizes that an adequate description of instream habitat condition is needed for successful interpretation of triad results espoused in Figure 2.12. Ecological measures of habitat condition usually involve rapid biological assessment protocols (RBPs) (Karr *et al*, 1986; Rankin, 1989; USEPA, 1989). Two examples, USEPA's RBP and the Ohio QHEI (Qualitative Habitat Evaluation Index) are provided in Table 2.35 that compare the two respective habitat index systems. Also presented is the approach taken by Richards and Host (Richards *et al*, 1994b). Indexes in Table 2.36 compare metrics of biodiversity that might typically be supported by the various qualitative habitat assessments outlined in Table 2.35.

Despite the focused investigation of RBPs, the methods in general construct a set of scoring indices that are comparatively qualitative and subjective in their attempt to describe bank and instream site conditions. Found in some form in Table 2.35, testing and quantitative measures of physical instream habitat such as particle size distribution, porosity, permeability, and stream velocity may be eventually necessary to distinguish between physical and chemical impairment in aquatic systems (Beebe, 1996; Belanger *et al*, 1985; Burton, 1992; DeWitt *et al*, 1988; Richards *et al*, 1994a; Sprague, 1985; Shirazi *et al*, 1981). Spatial scale and regional and local geology also play significant roles in determining the distribution of aquatic communities (Nelson *et al*, 1992; Richards *et al*, 1993, 1994b, 1996; USEPA, 1980, 1989).

2.20.8.1 Riparian and Instream Survey (RIS) Design

Habitat assessment has more recently focused on directly evaluating the physical influences of medium/fine sands, silts, and clays on community level macro-invertebrate diversity (Hawkins *et al*, 1990; Luedtke *et al*, 1976; Miller, 1988; Nelson *et al*, 1992, Richards *et al*, 1992, 1994b; USEPA 1995). Important mechanisms of impairment, substrate and cobble embeddedness are characterized by threshold impairment levels of fine materials < 2mm, which are physically dependent upon erosional features of the watershed, along with instream hydraulics (Diplas, 1994; Richards *et al*, 1994a, 1994b; Shirazi *et al*, 1981).

Table 2.35 - QHEI, USEPA RBP, and Richards' and Host's Approaches to Qualitative Habitat Assessment

EPA RPB Qualitative Field Habitat Scoring Indices (% weighting)	EPA RBA Associated Field Survey Metrics	Ohio QHEI Qualitative Habitat Scoring Indices (% weighting)	Ohio QHEI Associated Field Survey Metrics	(Richards, et al , 1994b) Mixed Habitat Metrics
Instream:	***Instream:***	***Instream:***	***Instream:***	***Instream:***
1. Substrate &	1. % Physical Substrate:	1. Substrate (20%)	1. Pollution Impacts	1. Substrate Composition
Available Cover (15%)	Bedrock, Boulder, Cobble	a. Type	2. Water Clarity	2. % Embeddedness
2. Embeddedness (15%)	Gravel,Sand,Silt,Clay	b. Quality	***Physical:***	3. Woody Debris
3. Flow/Velocity (15%)	2. % Organic Substrate	2. Instream Cover (20%)	3. Samp. Pass Distance*	4. Algae Abundance
Physical:	a. Detritus (CPOM)	a. Type	4. Water Stage*	***Physical:***
4. Channel Alteration (11%)	b. Muck-Mud (FPOM)	b. Quality	5. Field Gradient Est*	5. Per Reach Sampled*
5. Bottom Scour & Dep (11%)	c. Marl (Shell)	***Physical:***	6. Stream Section Length*	a. % Pool
6. Pool/Riffle	3. Run Velocity	3. Morphology (20%)	7. X-Section (Avg/Max):	b. % Riffle
Run/Bend Ratio (11%)	4. Land Use	a. Sinuosity	a. Width Profiles	c. % Run
Structural:	5. Local NPS Pollution	b. Development	b. Depth Profiles	6. Max Pool Depth*
7. Bank Stability (7.5%)	6. Sediment Odors	c. Channelization	8. X-Section % of Flood Stage:	7. Sinuosity
8. BankVegetation (7.5%)	7. Sediment Oils	d. Stability	a. Bank	8. Width*
9. Streamside Cover (7.5%)	8. Sediment Deposits	4. Pool Quality (20%)	b. Stream Bottom	9. Depth*
	9. Anerobic Conditions	a. Max Depth	c. Flood Plain	10. Flood Width*
	Physical:	b. Current	9. Bed Form Type*:	11. Stream Length*
Index is reach based.	10. Avg Stream Width*	c. Morphology	Run/Riffle/Glide/Pool	***Structural:***
	11. Avg Stream Depth*:	5. Riffle Quality (8%)		12. % Shading
	a. Pool	a. Depth	***Structural:***	
	b. Run	b. Substrate Stability	10. Canopy - % Open	
Note (*) quantitative metric	c. Riffle	c. Sub. Embeddedness	Sampling Path	
otherwise, considered	12. High Water Mark*	6. Map Gradient (10%)		
primarily a qualitative metric	13. Dam Present	***Structural:***		
	14. Channelization	7. Riparian Zone (10%)		
	Structural:	a. Width		
	15. Rel. Canopy Cover	b. Quality		
	16. Erosion	c. Bank Erosion		
			<-- *4b & 5c of QHEI score*	
		Index is site based.	<-- are considered instream	

Reach-level data based on USEPA's Rapid Bioassessment Protocol (USEPA, 1989). Site-level data based on Ohio EPA's Qualitative Habitat Evaluation Index for Fish (QHEI) (Rankin, 1989). Richards and Host executed list on 200m reaches at 11 gravel bed sites above a common lake (Osborne et al, 1991; Richards et al, 1994b). The reach-level distinction is loosely held based on increased reach lengths commonly incorporated for macro-scale (physical and structural) parameters extending upstream beyond a single riffle-run-pool habitat sequence.

Table 2.36 – Ohio EPA, USEPA RBP Level II, and Richards' and Host's Benthic Macroinvertebrate Assessments

EPA RPB Level II Qualitative Field Biological Condition Scoring (% weighting)	**Ohio Invertebrate Community Index (ICI) Macroinvertebrate Community Condition (% weighting)**	**(Richards, et al , 1994b) Total and Functional Biological Indices Utilized**
	Substrate Sample Data:	
1. Taxa Richness 2. Family Biotic Index 3. Ratio Filterers/Scrapers Collectors 4. Ratio of EPT and Chiranomid Abundances 5. % Contribution of Dominant Family 6. EPT Index 7. Community Loss Index 8. Ratio of Shedders t/Total	1. Total # Taxa 2. Total # Mayfly Taxa 3. Total # Caddisfly Taxa 4. Total # Dipteran Taxa 5. % Mayflies 6. % Caddisflies 7. % Tribe Tanytarsini Midges 8. % Other Dipterans/Non-Insects 9. % Tolerant Organisms	1. Taxa Richness 2. EPT Taxa 3. Dipteran Taxa 4. Other Taxa 5. Filterers 6. Gatherers 7. Predators 8. Scrapers 9. Shredders
	Qualitative Abundance:	
Abundance: 1. CPOM Total 2. Riffle/Run Composite Total 3. CPOM + Riffle Run Total *Level II: Riffle/Run sample and CPOM sample. Index is reach based if multiple riffle/run habitats are sampled*	Abundance: 1. Total Qualitative EPT Taxa *Index is typically site based,incoporating single nearby riffle-run-pool habitat.*	

Ohio EPA sampling protocols were used in this work. USEPA site/reach-level protocol based on USEPA's Level II Rapid Bioassessment Protocol (USEPA, 1989). Site-level data based on Ohio EPA's Qualitative Habitat Evaluation Index (QHEI) (Rankin, 1989). Richards and Host executed 200m reaches at 11 gravel bed sites above a common lake (Osborne et al, 1991; Richards et al, 1994b). The reach-level distinction is loosely held based on increased reach lengths that can accompany expanded sampling of multiple habitats, for example sampling slow and swift moving riffle-run habitats each in defining biometrics for a single station. (CPOM- Course Particulate Matter)

A watershed-scale ecological risk assessment is significantly enhanced by an informed instream/riparian survey design which can be quantitatively related to watershed attribute databases for chemistry, biology, ecology, toxicology, hydrology, geology, and ultimately socio-economy. Several convergent methodologies for conducting large-scale instream and riparian surveys have developed over the last two decades (Karr, 1991; OEPA, 1989; Rankin, 1989; Richards *et al*, 1994b; USEPA, 1989). Most have in some way attempted to facilitate a more informed and sometimes quantitative based ecological assessment of habitat structure. It is clear that a quantitative approach is not necessarily advocated nor is over-emphasis of the importance of habitat in determining biotic integrity where the term “supportive” is operative (Karr *et al*, 1986, 1991; Yoder *et al*, 1996).

Methodologies presented by Ohio EPA’s QHEI score, to be used in this study, and USEPA’s RBP, attempt to integrate multi-disciplinary assessment measurements and associated endpoints commonly used in watershed ecological risk assessment. One interesting difference between USEPA’s habitat assessment approach and the QHEI score is that the QHEI is more of a site based score, where the USEPA analysis is more of a reach based assessment. Also shown in Table 2.35 are associated field parameters that may compliment respective evaluations of habitat.

Final field survey endpoints chosen for stream sampling and observation can likely incorporate strategies involving measurement of parameters relating various aspects of:

- Hydrology,
- Hydraulics,
- Morphology,
- Instream and bank sediment quality,
- Riparian type and quality,
- Land use and coverage, and
- Water quality.

2.20.8.2 RIS Parallels to Historical Rapid Bioassessment Techniques

A more intensive riparian and instream survey (RIS) can be developed as a field tool to be used within a broader scope Geographical Information System (GIS) (Aronoff, 1993) and watershed assessment approach. To some level, Richards’ and Host’s approach in Table 2.35 represented such an effort, expanding upon RBP data with land use/cover analysis.

Endpoints found in rapid bioassessment protocols (RBPs) for aquatic habitat have historically been broken down into three main sub-groups: physical, instream, and structural features (Rankin, 1989; USEPA, 1989), shown in Table 2.35. It is of interest to note that elements defined as structural all relate to bank stability, giving its due recognition as an indicator of system stresses. One conceptual way an expanded survey design can build upon these definition sets is through further segmentation into four field components addressing riparian structure, ecologically significant points of interest, instream hydraulic and morphological properties, and riparian buffer zone land use/cover identification.

Currently, relationships between many physically based watershed parameters (e.g. sediment/contaminant loading, particle size, hydraulics, land use/cover etc.), and often qualitatively based endpoints in rapid biological assessment protocols and bioassays, are not well understood. Emphasis in quantitative survey design needs can be placed on identification of an appropriate set of physically based riparian and instream measurements which can be more meaningfully related to ecologically based habitat assessment endpoints used in RBPs.

Habitat, as shown in Figure 2.12, may often need to be evaluated in the context of site specific considerations addressing the interaction and distribution of energy, water quality-quantity, sediment quality-quantity, and biota (USEPA, 1996c). Standard elements incorporated in many hydrologic-based sediment/contaminant models should perhaps be addressed. USEPA RPB elements covered address nine (9) instream, physical, and structural endpoints used in macro-invertebrate habitat assessment (USEPA, 1989). Elements highlighted below are also presented in Table 2.35.

- **Instream features include**:
  - Substrate cover,
  - Embeddedness, and
  - Flow/velocity.

- **Structural properties include:**
  - Bank stability,
  - Bank vegetation, and
  - Streamside covers.

- **Physical features include:**
  - Bottom scour and deposition,
  - Channel alteration, and
  - Pool/riffle-run/bend ratio.

The RIS survey design example given in Tables 2.37 (fine-scale) and 2.38 (coarser-scale) were based in part on earlier work completed by Delong and other researchers (Anderson *et al*, 1976; Delong *et al*, 1991; Richards *et al*, 1994b). Delong's work presented a workable segmentation of riparian features for field observation. This work can be expanded upon to better define relationships between riparian ecology and historical land use/cover definitions presented by Anderson. For actual use in riparian evaluations, Table 2.37 presents a synopsis of those field methodologies where Delong's original approach on riparian survey was successfully field tested and found to be practical. Riparian and bank slope metrics were major parts of Delong's approach along with use of riparian width and height metrics.

As another example application that could be used for Leading Creek, Anderson's and Richard's work can be modified to provide for a single sheet of Level IV codes for use in riparian zone field identification, shown in Table 2.38. Both Tables 2.37 and 2.38 were designed especially for evaluation of agricultural and abandoned mine land impacts and were actually applied in Leading Creek. Each bank side was evaluated separately as appropriate.

Table 2.37 – Instream and Riparian Survey Design; Fine-Scale and Instream Land Use/Cover Code Designations – Example Application

Instream and Riparian Vegetative Zone Survey Riparian Buffer Zone and Adjacent Buffer Fine Scale Regional Attribute Vegetative Cover Codes and Degree/Length/Locational/Ecology Codes	**Instream and Riparian Ecological Survey Point Attribute Instream Land Use/Cover Designation Codes**
Modified from (Anderson et al, 1976;Delong, et al, 1991) **Riparian Vegetative Land Cover Code:** 1 No vegetation (Barren Land --> Anderson Code 7) 2. Annual herbaceous crop (311) 3. Perennial herbaceous crop (312) 4. Herbaceous wild (313) 5. Herbaceous wild/shrubs (shrubs comprising 20-60% riparian veg.) (33) 6. Shrubs (32) 7. Trees (4) 8. Shrubs and trees (>1/3) (34) 9. Trees and shrubs (>1/3) (44) 10. Water willow (62)	**Instream Land Use/Cover Code:** (a) - Inflow (51): 1. - NPDES effluent discharge 2. - Direct acid mine drainage/seep to watercourse 3. - Other wastewater discharge 4 - Stormwater discharge (pipe) 5 - Stormwater discharge (unlined ditch) 6 - Stormwater discharge (lined ditch) 7 - Tributary inflow confluence (perennial) 8 - Tributary inflow confluence (perennial) with VAMD 9 - Tributary inflow confluence (non-perennial) 10 - Tributary inflow confluence (non-perennial) with VAMD 11 - Pond/lake/reservoir inflow confluence *VAMD - Visual AMD impact present at this location*
Slope Angle Rating (degrees) 1. Undercut 2. 71-90 3. 51-70 4. 31-50 5. 10-30 6. 0-9	*(b). - Downstream Flow Obstruction (17):* 1. - Fence/flap obstruction 2. - Man-made dam 3. - Beaver dam 4. - Significant channelization
Length Rating (feet) 1. None (0) 2. 0 - 3 3. 3 - 9 4. 9 - 15 5. 15 - 25 6. > 25	*(c) - Stream Channel Crossing Impact (17):* 1. - Exposed petroleum line crossing 2. - Exposed natural gas line crossing 3. - Exposed sanitary sewer line crossing 4. - Other urban or built-up land use code 5. - Farm animal crossing 6. - Farm tractor crossing 7. - Automobile/truck forded crossing
Erosion Origin Code 1. Ungulate - Natural 2. Agricultural 3. Acid Mine Drainage 4. Other **Instream Locational Code** R - Right Edge of Water (looking DS) L - Left Edge of Water (looking DS) Cen - Channel Center **Ecological Indicator Code** 1. Tracks 2. Stools 3. Visual siting 4. Other **Ecological Population Code** 1. Aquatic macrophytes 2. Reptiles 3. Amphibians 4. Aquatic birds 5. Mammals	*(d). -Potentially Impacted Instream Sediment/Waters (173)* 1. - Abnormal sediment color 2. - Sediment oils/slicks 3. - Sediment odors 4. - Abnormal water color 5. - Water oils/slicks 6. - Water odors 7. - Foam/Surfactants 8. - Eutrophication *All codes subject to change*

Table 2.38 – Instream and Riparian Survey Design; Land Use/Cover Code Designations – (Coarser-Scale) - Example Application

Land Use/Cover Designations and
Associated Modified Anderson Category Codes
Base Scale (1:24000)
Instream & Riparian Zone Survey and General Application
Leading Creek Improvement Plan

Modified from (Anderson,et al, 1976)

Land Use/Cover Special Designations:	**Land Use (Continued):**
10 - No Observation	
100 - Not Applicable	***2 -Agricultural Land***
	21 - Crop Land & Pasture
Land Use:	211 - Crop Land
	2111 - Cultivated Crop Fields
1 -Urban/Built-Up Land	2112 - Idle Crop Land
111 - Residential - low density	212 - Pasture
112 - Residential - medium density	2111 - Cultivated Hayfields Fields
113 - Residential - high density	2122 - Confined Pasture wrt RBZ
114 - Residential mobile home community	2123 - Unconfined Pasture wrt RBZ
	2124 - Idle Pasture
12 - Commercial and services	23 - Confined Feeding Operations
121 - Institutional/government	231 - Confined Feeding Operations wrt RBZ
122 - Automobile/truck service/fueling	232 - Unconfined Feeding Operations wrt RBZ
123 - Recreational areas	24 - Other Agricultural Land
1231 - Natural use area	
1232 - Developed use area	**Land Cover:**
124 - Environmental Monitoring Station	
	3 -Rangeland
13 - Industrial	31 - Herbaceous rangeland
131 - Manufacturing	32 - Shrub and brush rangeland
132 - Wood processing	33 - Mixed rangeland
133 - Petroleum extraction/storage	34 - Mixed rangeland and forest
134 - Petroleum storage	
135 - Natural gas extraction/storage	***4 -Forestland***
136 - Salvage/junk yards	41 - Deciduous forest
	42 - Evergreen forest
16 - Mixed urban or built-up land	43 - Mixed forest
	44 - Mixed forest and rangeland
17 - Other urban or built-up land	
171 - Graveyards	**5 -Water**
172 - Raw materials	51 - Streams and canals
1721 - Road salt piles	511 - Perennial streams
1722 - Coal piles	512 - Non-perennial streams
1723- Gravel/sand/soil piles	513 - Drainage ditches and canals
173 - Abaondoned/waste materials	52 - Ponds/lakes
1731 - Mining surface spoils	53 - Reservoirs
1732 - Tire piles	
1733 - White goods	***6 -Wetlands***
1734 - Junk automobiles/scrap metal	61 -Forested wetlands
1735 - Household waste/garbage	62 -Non-forested wetlands
1736 - Sanitary/solid waste landfill/pile	
1737 - Construction/demolition/debris fill/pile	***7 -Barren lands***
1738 - Hazardous materials/waste fill/pile	74 - Bare exposed rock
	751 - Surface mines
	752 - Quarries and gravel pits
	76 - Transitional areas
All Level III & IV codes subject to change	761 - Disturbed land
Level I & II will be based on Anderson definitions	7611 - Construction/development
RBZ - Riparian Buffer Zone	7612 - Logging
	7613 - Other

2.20.8.3 Contemporary Habitat Bioassessment Techniques

As a more recent update of the integration of these methodologies in habitat structure assessment, USEPA underscores that habitat assessment provides information on both habitat quality and defines constraints on the site's potential to reach use-attainment. The habitat assessments are critical to any evaluation of environmental integrity (Karr, 1991; USEPA, 1996c). Integrated habitat assessments today follow familiar approaches but may also incorporate additional description compared to Table 2.35. USEPA gives several attributes that may serve as a typical framework for contemporary habitat assessment (USEPA, 1996c).

Contemporary instream, structural, and physical designations below were added to maintain comparison with the outline listed above:

- **Instream features amended to include**:
  - Substrate variety / instream cover
  - Bottom substrate cover,
  - Embeddedness, and
  - Flow or velocity / water depth.

- **Structural properties amended to include:**
  - Upper bank stability,
  - Lower bank channel capacity,
  - Bank vegetation stability,
  - Riparian vegetative zone width, and
  - Canopy cover (shading).

- **Physical features amended to include:**
  - Bottom scour and deposition,
  - Channel alteration,
  - Pool/riffle run/bend ratio, and
  - Channel sinuosity.

USEPA lists a candidate set of variables that may be useful in evaluating environmental conditions and their relationships to both geographic scales and the five environmental factors subject to human influence. These are similar to other descriptions given here but are used to separate variables based on scales. In increasing order of complexity, ecological scales are given by (USEPA, 1996c):

(a) Watershed,
(b) Riparian and bank structure,
(c) Channel morphology, and
(d) Instream.

2.21 HYDROLOGIC SIMULATION MODELS

Hydrologic simulation models represent a distinct group of computer based watershed modeling tools available to the hydrologist and water resources engineer. Simulation models represent a class of flow models that can include address of groundwater, surface water, and other aspects of the hydrologic cycle like evapotranspiration (Fetter, 1994). Simulation models relate rainfall patterns to run-off patterns (Linsley *et al*, 1975). Runoff is primarily driven by rainfall intensity and duration, and will be a function of the watershed's size, slope, shape, storage, morphology, channel-types, soil types, and percent impervious land (Bedient *et al*, 1992; Bicknell *et al*, 1993).

Hydrologic simulation models have been classified for watersheds and cover a wide array of approaches and characteristics (Bedient *et al*, 1992). They can include model types and model examples depicted in Table 2.39.

Table 2.39 - Hydrologic Simulation Models
(Adapted from Bedient *et al*, 1992)

Model Type	Example of Model
Lumped Parameter	Snyder Unit Hydrograph
Distributed	Kinematic wave
Event	HEC-1, SWMM
Continuous	Stanford Watershed Model, SWMM, HSPF, STORM
Physically Based	HEC-1, SWMM, HSPF, SWRRB
Stochastic	Synthetic stream flows
Numerical	Explicit kinematic wave
Analytical	NASH IUH

The more complex models such as HSPF, SWRRB and SWMM have an underlying physical basis, for example working on the principles of continuity and conservation of mass and momentum (Arnold *et al*, 1995; Donigian *et al*, 1995). These models all attempt to simulate hydrology of the system in some way, predicting among other things, water flow on some incremental time step basis. A simulation is at some level just a series of static, discrete events. In hydrologic simulations, when dealing with several streams or reaches, a concept of flow routing must also be applied between network elements. For watershed analysis, the parallels are normally drawn between use of lumped parameter versus distributed parameter models, event versus continuous models, and stochastic versus deterministic models (Bedient *et al*, 1992). In hydrologic simulations for HSPF, when dealing with several streams or reaches, a concept of flow routing must also be applied between network elements

For rainfall-runoff analysis, hydrologic simulation time steps include perhaps daily, hourly, 15 minute, and even 5-minute time steps. Simulation in this sense infers a transient component, with transient inputs (Bicknell *et al*, 1993). The idea of the time-simulation model is that a time-sequence input data set is used to drive the model (e.g. using rainfall), which then predicts a time sequence output data set representing some parameter of interest in the watershed (e.g. flow rate or stream stage). As another example, using one of the models listed in Table

2.39, a hydrologic time-simulation of the New River might for instance predict 30 years of daily stream flow data using a 30-year input record of daily rainfall data. In this case hourly input data would be preferable if reasonable accuracy on a daily basis was needed.

Both sediment and water flow can be included as output variables of the simulation model, as well as loading of other pollutants depending on the actual simulation model chosen. Watershed models may range from simplified to complicated 1-D representations of stream networks based on Manning's equation (HEC and HSPF), to more sophisticated attempts at solving St. Venant's equations for 3-D water flow (DWOPER) (Bedient *et al*, 1992). For example HSPF is a 1-Dimensional (1-D) flow routing model, based on uniform flow assumptions, a fair assumption for most alluvial streams. The model though requires a great deal of input data and initial set-up time. Whichever model is chosen, a thorough approach to river engineering will want to address two phases of hydrologic flow, including water and sediment (Yang, 1996).

A major advantage of simulation models is the insight gathered from the shear detail and labor that goes into compiling such a complex model application (Bedient *et al*, 1992). This implies that one will gain a fair amount of insight simply by reviewing the watershed in this manner. Another distinct advantage, the main advantage of simulation models, is that once set-up, they can be used to quickly and efficiently test various management schemes (e.g. reduced loading scenarios). They are readily able to evaluate landscape-level impacts upon the quality and quantity of water (Bedient *et al*, 1992; Linsley *et al*, 1975). Drawbacks to simulation models include the amount of data needed to achieve valid results where many applications of simulation models are not properly calibrated and verified. Model accuracy is largely determined by model input (Bedient *et al*, 1992; Bicknell *et al*, 1993; Linsley *et al*, 1975).

2.21.1 Simulation of the Landscape: PS and NPS Modeling

While some of the models listed in Table 2.39 look at stormwater issues only, some continuous simulation models also retain the ability to evaluate both non-point source (NPS) and point source loading influences together (HSPF, SWRRB). These can be addressed along with properly incorporating tributary inflows, all of which may change with time (i.e. transient, steady-flow). Because of needs to better distinguish specific sources of stress in a watershed, a more in-depth quantitative simulation approach to hydrologic modeling is often needed, such as use of HSPF or SWRRB.

When all water/sediment quality parameters of potential concern may not yet be identified, a robust model will be able to better incorporate generalized or specific chemical constituents as the investigation proceeds, such as HSPF. This may become necessary to adequately describe sediment and contaminant loading impacts in a complex stream system. For some simulation models, expected potential parameters of concern might realistically include sediment, pesticides, heavy metals, and potentially other synthetic organic materials. All of these pollutants including flow can be simulated to some degree by developing in-depth knowledge of the landscape's hydrology (Arnold *et al*, 1995; Bicknell *et al*, 1993; Donigian *et al*, 1995).

Minimum flow/transport technical simulation model criteria might include the following abilities (Bicknell *et al*, 1993; USEPA, 1988, 1991):

- Ability to provide watershed/basin-scale analysis of rainfall/run-off patterns for high, medium, and low flow stream conditions,

- Ability to provide a calibrated flow model for use in sediment/contaminant fate and transport analysis,

- Ability to identify significant PS and NPS sources of sediment and contaminant loading in the stream system, and

- Ability to evaluate reduced sediment and contaminant loading scenarios from significant PS and NPS sources identified.

Simulation model input requirements are anticipated to include basic data and information usually describing the following in some key aspects of hydrology and hydraulics:

- Stream Discharge
- Precipitation
- Evapotranspiration
- Sedimentation (including bed/overland fractions of clay/silt/sand)
- Water Quality
- Land Use and Vegetative Cover

2.22 GEOGRAPHICAL INFORMATION SYSTEMS (GIS)

Geographical Information Systems (GIS) are quickly becoming ubiquitous tools in watershed management applications, along with having increased focuses in areas of science and engineering. GIS is used extensively in managing the input data for simulation models described earlier. This is because visual and internal strengths of GIS in communicating and analyzing spatially distributed information. The definition of a GIS can cast a relatively wide net as given by the following (Aronoff, 1993):

> "A *geographic information system is any manual or computer based set of procedures used to store and manipulate geographically referenced data*"

Distilled down to a more workable definition of computer based tools, a geographic information system can also be defined as a computer based system that provides the following four capabilities to handle geographically referenced data (Aronoff, 1993):

1. Input
2. Data management (data storage and retrieval),
3. Manipulation and analysis, and
4. Output.

According to Aronoff's definitions, in this classical breakdown of GIS components, data input is the greatest cost of most GIS systems. Data management represents the process by which the GIS stores and retrieves data from the underlying database and data manipulation and analysis indicates what types of data can actually be generated by the GIS. The spatial analysis capabilities afforded by the GIS distinguish this class of computer based tools from other graphics oriented systems like computer aided design and drafting (Aronoff, 1993).

As an example in watershed management, a simulation model is often desired that allows integration of flow and transport model simulation results with an integrated GIS based spatial data decision system (SDSS). Ideally it is desired to exploit the data management capacity of a GIS to help generate the input files for simulation model runs (USEPA, 1981).

2.22.1 Examples of GIS Applications

A common example of applying a GIS in environmental problems is using the GIS to develop areal estimates of various properties describing land use and cover characteristics. For example %Bare Soil in each subshed, or the %Strip Mines, or the percentage of impervious land. Defined above, the GIS can be used to input, manage, store, and represent spatial data for watershed management projects and programs. It is a naturally self-focused tool because of its ability to exploit many different scales of input information available, and to incorporate spatial data into thematic maps. As indicated in the basic description above, the GIS retains powerful capabilities in cross-referencing data, aggregating data, combing data, and otherwise analyzing spatial data statistically. Data analysis capabilities can sometimes be overlooked by GIS-users where map generation capacities of a GIS can be over emphasized.

Among the standard analysis and computer presentation features of GIS, the benefits of the GIS can also provide useful maps for field surveys. These can include creation of useful buffer zones calculated around each reach investigated, for example applied to a riparian and instream survey design. Geodata can be useful in pointing the way to remote site access. GIS maps as field tools give a quick understanding of the layout of landscape that can assist both field notation and data screening techniques in watershed investigations. Provided they have inherently accurate and useful information, GIS mapping offers immediate field recognition of local plan-form geomorphology and land use/cover at sites chosen for a given survey. Buffer zones created can serve as a spatial ruler in the field, making estimation of distances easier during data collection missions. In combination with geographical positioning systems (GPS), GIS and GPS together will continue to move to the forefront of field survey equipment combinations used by future researchers and land managers of watershed environments.

In summary, the primary function of GIS is spatial data management. GIS though also possesses inherent and unique analysis and mapping capabilities that draws many users. While the mapping aspect of a GIS is well known to most people familiar with the term, the underlying structure and analytical power of GIS is less known and less developed in applications. There may also be a misconception sometimes that a GIS is more than a tool (Aronoff, 1993). This is changing with more and more integration of standard simulation models, GIS, and expert system approaches designed for watershed management such as BASINS, HSPF, and ARCView combinations for instance (Donigian *et al*, 1995). Examples of GIS applications are numerous and include agriculture and land use planning, forestry and wildlife management, archaeology, geology, municipal applications, and ecosystem management (Aronoff, 1993). GIS is useful to almost every walk of life that deals with managing spatially distributed data.

2.22.2 Limits of Current Digital Data Sets

Reported by Singh (1995), the status of existing geodata sets for coverages such as soils, geology, land use, or vegetative cover may suffer from the following inherent traits:

- The geodata sets are in different formats,
- The geodata sets use different coordinate (projection) systems, and
- The data is too coarse for 2-D or 3-D watershed models.

For the last aspect noted above, the issue of data resolution can also be a limiting issue in many 1-D model applications also, again dependent on the quality of the input data sets. As time goes on, for watershed management applications GIS and simulation models will be more difficult to discuss exclusively of each other. Fundamentally, each is a tool that must be integrated by the user with the application of expertise. This will occur in a formal or informal manner.

2.22.3 Relating GIS and Benthic Macroinvertebrate

As instructional examples, Richards and Host (Richards *et al*, 1994b, 1996) developed important works looking into the utility of modeling biological response of the system from the perspective of watershed variables and GIS processing. The techniques involved by and large standard data collection methods and utilized coarse-scale land use/cover geodata sets. The second 1996 paper follows up the original GIS discussion and the same basic approach to habitat assessment, but added elevation and other geologic-hydrological land factors with improved success. The second work covered 45 sites in Michigan where the original investigation looked at 11 of these.

Richards' and Host's original 1994 land use and cover data was broken down by categories including a) Urban, b) Agricultural, c) Deciduous Forest, d) Mixed Forest, and e) Lakes and Wetlands. Their GIS was complied at a scale of 1:250,000 series (16ha resolution), with 1:100,000 drainage analysis and 1:24,000 housing density. No drainage areas were reported for the 11 basins studied in 1994, but based on reported scale, mapping, and stream lengths, the streams investigated appeared to be 2nd or 3rd order, 3.5 to 18km in length, much like Leading Creek's midsize tributaries. The 11 streams originally studied by Richards and Host were gravel and sampled along 200m reaches, 2 to 3km above a common lake.

Important aspects of Richards' and Host's 1994 and 1996 work were that they initiated a fruitful discussion along the lines of potentially using land use and vegetative cover data as well as other field data collected to predict biodiversity metrics instream. A significant result from their initial efforts was that it appeared that there were useful relationships to be gleaned from data on landscape and regional scales, but finer scales of perhaps 1:24000 may be needed to accurately predict habitat or biodiversity instream, on a reach by reach basis. Other summary comments and qualifications of their work are included below (Richards *et al*, 1994b, 1996).

- No sediment chemistry, water chemistry, or toxicity studies appeared to accompany evaluations of biodiversity.

- ♦ Richards and Host did not complete the full rapid bioassessment protocol by USEPA, but used a similar set of habitat metrics shown in Table 2.35.

- ♦ A shorter list of qualitative and quantitative indexes similar in many regards to the RBPs was used with added emphasis given to fluvial morphology. For instance embeddedness and substrate were used, but less attention was paid to physical riparian characteristics and no field survey of buffer zone land/use cover was made.

- ♦ Richards and Host added an additional focus of evaluating a qualitative index for woody debris, somewhat already functional in the 2-part substrate index they used for substrate and instream cover RBP subscores. The latter subscore mentioned for example is an organic matter metric (OEPA, 1989; USEPA, 1989).

- ♦ In their approach they created a 100m riparian buffer zone along stream lengths, filled it with the 1:250000 land use/cover data, and, except for wetlands, concluded that little differences (<5%) existed between upstream watershed-scale data and riparian zone scale data. They proceeded with analysis of watershed-scale land use/cover variables only (based on upstream % by area).

- ♦ Richards and Host completed their macroinvertebrate sampling similarly to methods used on Leading Creek.

- ♦ Richards and Host did not evaluate land or instream flow and sediment hydraulics.

- ♦ Their data analysis approach was to first pick a subset of the 15 habitat survey variables via strength of Principal Components Analysis (PCA) for biodiversity. They then attempted to associate those habitat variables by Pearson correlation to GIS watershed variables.

- ♦ Richards and Host found that 8 of 15 habitat variables had significant correlation with 9 biodiversity indexes (1st 4 axis = 75.3% of total biodiversity data variation [P]<0.05 and 0.10). For axis I = 35.1% of total variation, dominant biodiversity featured was species richness. For axis II (17.1%) filterers were the dominant biodiversity metric exhibited. Dominating axis I habitat features were embeddedness, substrate size, and woody debris. The other five were algae, stream width, sinuosity, %run, and %shade.

- ♦ Five of the above eight variables mentioned had significant habitat variables also correlated ($p<0.05$) with some watershed variable. Embeddedness (+0.63 to +agriculture), %run (-0.61 to +urban), algae (+0.73 to +housing density and -0.68 to +mixed forest), %shade (+0.67 to +housing density), and stream width (+0.60 to link #). They found substrate size correlated to urbanization ($p<0.10$) (-0.55 to +urban).

- ♦ Richards and Host then focused back on Principal Component axes I and II, and concluded that embeddedness, substrate, woody debris, and algae were significantly correlated to land use/cover data sets. They could reasonably predict three of four variables using GIS land use/cover data, but were not able to predict woody debris accurately which they attributed to poor data resolution.

- Embeddedness was overall the strongest variable seen linking biodiversity to GIS variables. In general, embeddedness was explained mostly as variation in richness and filterers.

- The three worst biodiversity scores (sites) where associated with the highest embeddedness scores, all three having smaller gravel sizes. This could be consistent with increased trapping of fines and faster, shallower sealing of the subpavement.

- In the later work, partial redundancy analysis revealed that geologic and land-use variables had similar magnitudes of influence on stream habitats. Of the geologic variables, catchment area, proportion of lacustrine clays, and glacial outwash materials had the strongest influence on physical habitat, particularly on channel dimensions (Richards *et al*, 1996).

2.23 KNOWLEDGE-BASED EXPERT SYSTEMS

According to one definition system for artificial intelligence (Lemon et al, 1999; Newell, 1990):

- An expert system is defined as an artificial intelligence technique in which the knowledge to accomplish a particular task (or set of tasks) is encoded *a priori* from a human expert,

- An expert system consists of two pieces: the knowledge-base and the reasoner. The knowledge base represents a set of codified rules. The reasoner than exploits the knowledge (of the rules) to apply that knowledge to a particular problem.

- In this definition set by Newell, intelligence is defined as the degree to which a system approximates a knowledge-level system. A knowledge-level system is one that rationally brings to bear all of its knowledge into every problem it attempts to solve.

As implied above in the working definition for knowledge based expert system applications, examples of implemented expert systems all impart the delivery of advice to some user, through use of a computer interface. Expert systems can also be defined as an interactive computer program that encodes judgement, experience, rules of thumb, intuition, empirical knowledge, and other forms of information or expertise to give knowledgeable advice on a specified problem (Singh, 1995).

A knowledge-based expert system is one that gives us the best advice, based upon everything that it knows about the problem one defines for it. This best advice is almost always codified as a computer program that is used to interface with the user. Indicated by Singh, expert system applications have not quite found a niche yet in water resources or watershed modeling and hydrologic simulation (Singh, 1995). One of the first attempts at such an approach was HYDR. HYDRO was an expert system that evaluated physical characteristics of a watershed (Gaschning *et al*, 1981; Singh, 1995). The model attempted to apply an expert system rule base to develop data input for HSPF (Bicknell *et al*, 1993; Donigian *et al*, 1995).

2.23.1 Other Examples of Expert System Implementation

Examples of implemented expert systems outside the ecological and hydrological modeling realm include (Lemon et al, 1999):

- MYCIN: Diagnosis of Infectious Diseases
- MOLE: Disease Diagnosis
- PROSPECTOR: Mineral Exploration Advice
- DESIGN ADVISOR: Silicon Chip Design Advice
- R1: Computer Configuration

One example of an expert system application was made in malting barley crop management (Broner *et al*, 1996). For the barley expert system developed, two major drawbacks of using expert systems were identified including (a) human expert advice is not always reliable and environmentally sound, and (b) adjusting the expert system to site specific conditions is time-consuming and difficult to automate. This work pointed out the limitations of an expert system. GYPSES is another decision support system for the management of gypsy moths using a knowledge-based geographic information system and multiple knowledge-based modules (Twery *et al*, 1993). In the Woodland Evaluation System, a simple expert system provided a framework for evaluating the conservation, landscape and recreation aspects of woodlands in southern Scotland (Edwards-Jones *et al*, 1996).

An expert system EXPRES (EXpert System for Pesticide Regulatory Evaluation Simulations) was developed to aid users in the assessment of the potential for pesticides to contaminate groundwater (Crowe *et al*, 1992). An example of tool integration, in the EXPRES system, numerical models were used to simulate the transport and transformation of pesticides in the unsaturated zone, aiding the user in generation and understanding model output results. EXPRES is a knowledge-based system that characterizes the physical, climatic, hydrogeological, pedological and agricultural settings of typical agricultural regions across Canada. The system can be used for making policy decisions.

Many other expert systems are available and cover several important areas of risk management including seismic evaluations, radioactive waste handling, waste disposal site selection, and disaster mitigation. There are many applications of expert systems in ecological watershed management. One example included evaluating instream aquatic habitats where the work was also attempting to link these tools with stream flow simulations (Grenney *et al*, 1994).

2.24 COMBINING SIMULATION MODELS, GIS, AND EXPERT SYSTEMS

As described above, each of the tools noted has a special set of functions and purpose, but there are overlapping features. The most common feature between the tools is that they are all represented and managed in some way by a computer program. Simulation models, GIS, and expert systems all require some type of user input to function properly. From this viewpoint and the definition of artificial intelligence, all simulation and GIS tools might be considered in some formal sense to represent expert systems. While it is obviously possible to manage projects with each one of these tools separately, apart from the other two, it is also common to see all three tools utilized jointly. A GIS is normally going to be critical to developing accurate input data for watershed model simulations. An expert system in hydrology will likely depend in some way on simulation models, and in turn may depend heavily on GIS for data management.

From a more practical standpoint, simulation models and GIS are subordinate tools to the expert system. An expert system would be expected to interpret for example several different simulation "runs" or scenarios, and offer its best advice for example as to which scenario should be selected. The integrative nature of expert systems is shown in the EXPRES expert system, where all of these tools are feasibly incorporated together; e.g. simulation models, GIS, and an expert system. It was discussed previously how GIS and simulation models combine to form a complimentary approach to watershed modeling, using the GIS to both acquire input data for simulation runs, but also then to use the GIS to express model results in some meaningful way. For instance mapping maximum monthly load rates per unit area for different areas is a conceivable combination of using the GIS and simulation models together.

2.24.1 Applications of Simulation Models, GIS, and Expert Systems

Discussion thus far on simulation models, GIS, and expert systems were intended to also address a discussion of various examples of these tools and their applications to environmental management of non-point source pollution (NPS). The many examples given above in describing each tool show various types of either separate or joint application of the tool set applied to NPS management schemes.

A key aspect of resource management at watershed spatial scales is the need often to address NPS pollution together with any known point source (PS) pollution problems (USEPA, 1996c, 1998). This for example may include looking at agricultural impacts concurrent with strip mining influences, mixed with silviculture activities, and also perhaps having to simultaneously address sanitary and industrial wastewater discharges added into a stream. This is a typical profile for many watersheds found in the eastern United States. It presents typical dilemmas and questions to risk managers.

Because water is such a good solvent, water pollution has become a common denominator in the problems relating human influence to ecology. Because valued ecological components are also organized at the watershed-scale (Cairns *et al*, 1991, 1996; Vannote *et al*, 1980), we realize an increasing need to be able to predict the downstream impacts of man's activities on the landscape. We desire to understand impacts both locally and upon the watershed as a whole, e.g. holistically (NRC, 1992; USDA, 1998). Lately aquatic ecology has been shown to be a natural focus point for watershed resource inventory control. The idea is founded in assigning use attainment status based on biological and physical databases, for instance using ICI or IBI scores (Karr *et al*, 1981a, 1981b, 1985; Richards *et al*, 1992, 1994b, 1996; Rosenberg *et al*, 1986, 1993). GIS itself plays an increasingly important and more frequently utilized role in watershed ecological based investigations (Altman *et al*, 1993; Aronoff, 1993; Donigian *et al*, 1995; Richards *et al*, 1994b, 1996; Singh, 1995).

2.24.2 Developing Predictive Management Tools

Environmental engineers are concerned especially about water quality and quantity. As such engineers desire quantitative and qualitative assessment routines (algorithms) that enable reliable prediction of the key properties of water produced by any watershed. This in some sense summarizes the direction headed in understanding NPS influences that are distributed throughout the landscape (USEPA, 1996c). The relationship between landscape and biodiversity is defined by NPS stresses that communicate with ecology via water and sediment (Chapman *et al*, 1992). An expert prediction or assessment of quality or quantity of water and sediment can take many forms. For example predicting flood crest and time of crest, or maximum load rates of suspended sediment or various pollutants, or showing extension agents colored maps indicating highly erodible field conditions or prime farmland areas. These are all examples of how simulation models, GIS and expert systems are used today and how they may further converge in the future.

It is conceivable that for example biologists may have models in the near future that simulate animal populations over time, based on temperature and rainfall. The applications of GIS and expert systems are also of course not limited to problem statements of predicting only the quantity or quality of water and sediment, or ecology. Each tool can also be used for meeting many other kinds of ecological resource management goals associated with NPS management schemes. For example, these may include use of GIS maps to depict animal populations, land use/cover characteristics, and using expert systems to show state regulators how to develop site-specific treatment technology reviews and action plans for any given leaking UST (underground storage tank) site. There are a variety of ways in which simulation models, GIS, and expert systems can be conceived, developed, and ultimately linked. A focus on hydrology and applications of simulation models, GIS, and expert systems has been stressed here.

2.24.3 Integrating the Watershed Management Tools

Ecologists, engineers, land managers, and economists will eventually converge to a common language of ecological goods and services produced by watersheds (i.e. goods and services produced by nature). If curiosity does not dictate this pathway, economics will eventually demand it. From the point of view of environment and engineering, simulation models, GIS, and expert systems present useful tools. These tools help solve some portion of the primary question posed today; that of predicting the quantity and quality of water (and sediment) produced by any watershed. Of GIS, simulations, and expert systems, each addresses information management in a unique way, but with often overlapping scopes. GIS and simulation models are naturally coupled. Expert systems are partially inherent in the simplest approaches found today in watershed management. The latter are also more obviously portrayed in the most complex watershed assessment schemes and land management approaches seen.

Drawbacks to integrating expert systems with GIS on large watershed-scales for NPS management can include the complexity of the problems being solved, and the difficulty to automate such a process for widespread application. On the other hand the simple combination of utilization of Best Management Practices for example in conducting various pollutant loading scenarios with a simulation model, while using GIS, is a viable example of the benefits of integrating all tools together in one project (Gaschning *et al*, 1981; USEPA, 1981). The EXPRES project was given as an example three-tool approach that could be used to combine these elements successfully (Crowe *et al*, 1992).

All three tools have each found more and more prominence lately in watershed management schemes because of their proven utility in pollution prevention applications, reduced loading scenario modeling, and data management (Aronoff, 1993; Xinhao *et al*, 1997). The power of computer processing has for example made projects like USEPA's "Surf Your Watershed" and USEPA's BASINS Model possible. These are "integrative" in nature, and represent a glimpse into the future of watershed modeling, GIS data management, expert system applications, and, ultimately, ecosystem management. This future is quickly approaching.

2.25 POINT SOURCE AND NON-POINT SOURCE MODELING

Some continuous simulation models retain the ability to evaluate changes to both non-point source (NPS) and point source (PS) loading (Arnold *et al*, 1995; Donigian *et al*, 1995). Using simulation models, these can be addressed while properly incorporating tributary inflows, all of which can change with time (i.e. transient, steady-flow). Because of needs to better distinguish specific sources of stress in a watershed, a more in-depth quantitative simulation approach to hydrologic modeling is often needed, such as use of HSPF or the simpler basins model SWRRB.

The data requirements of HSPF and SWRRB are extensive and require input for each significant subshed of the watershed. Just the number of variables in HSPF would require approximately 40 document pages of typed material to list out, line by line. The various subsheds, representing a continuous downhill segmentation of the stream, are routed together using a time series manager. Each subshed is then simply defined by various downhill inflows and outflows from various parts of the watershed. The idea of both SWRRB and HSPF sediment transport functions are to track sand, silt, and clay as separate fractions coming off the hillsides and flowing into and down channels. Integrating multiple modeling approaches in a single platform, USEPA's BASINS package for example allows access internally to the models HSPF, TOXIROUTE, and QUAL2E (USEPA, 1996b).

2.25.1 Variations on Flow and Water Quality Models

Potential weaknesses of some integrative approaches are that many simulation models are only applicable at field-scales, do not incorporate sub-surface processes, or do not adequately address run-off, detailed land use/cover, sedimentation or water quality concerns, simultaneously. Complicated watershed-scale and basin-scale approaches would tend towards HSPF and SWRRB for developing sophisticated simulations that may want to emulate land use/cover patterns/changes, in an integrated GIS and expert system environment.

Biological models such as IFIM and FABISHIM may be suitable candidates for many model selection processes but not others based upon their inability to adequately simulate instream sedimentation and water quality processes throughout a basin (Bicknell *et al*, 1993; Donigian *et al*, 1995; Milhous, 1982; Wesche *et al*, 1980). The latter limitations may be considered serious drawbacks to the degree that the systems do not address key physical mechanisms associated with NPS stresses, such as sedimentation.

As it is with many hydrologic models, lack of calibration data associating biological condition and variable flow conditions in a watershed or basin is usually a severe limitation of biological simulation models or fisheries models. Such calibration data may not be feasible to collect given levels of impairment, variation of impairment, and size of a basin. As an example, without an ability to holistically address upstream sedimentation issues interactively, the validity of applying these types of expert model systems to integrate and then evaluate all NPS influences in a watershed does not appear feasible, though field-scale applications can be successful.

In some respects many may view the biological aspect of the model FABISHIM to represent an encoded expertise for example. Thus there are successful strategies which can incorporate knowledge of ecology and hydrology in a three-tool approach, with limitations noted. A major limitation today, lack of adequate calibration data and post validation data can limit the joint application of simulation models, GIS and expert system approaches.

2.25.2 The Model SWRRB

SWRRB (Arnold *et al*, 1990, 1995) is a less sophisticated model then HSPF where SWRRB is particularly useful for large basin analysis and simulation. SWRRB concentrates on long term weather, hydrology, and sediment yield of large basins. An advantage is it can be used in ungaged watersheds (presumably this simply means it can be used without calibration). It is limited to 5 sub-basins normally in differentiating upstream influences, with 10 soil layer descriptions possible. Subbasins can be expanded but all are apparently routed to the basin outlet, on a daily basis. SWRRB is not well suited for flood control analysis or routing. SWRRB does not allow user defined constituents and does not handle complex water quality simulation in streams. Recent work on the model though does include a pesticide fate component GLEAMS, optional SCS Curve technology for estimating peak run-off rates, and newly developed sediment yield equations (Arnold *et al*, 1995). SWRRB can handle nitrogen and phosphorous nutrient cycling.

Constraints of SWRRB's routines may become limiting in some applications, considering HSPF capabilities, though recognizing the data input requirements increase with HSPF. The sediment wash-off routines used in SWRRB are based upon the Modified Universal Soil Loss Equation. The MUSLE is less sensitive than the routines in HSPF, which are based on the 1976 ARM and NPS models. A highly segmented basin, with a high degree of land-use resolution may require HSPF over SWRRB (Donigian *et al*, 1995; Linsley *et al*, 1975; USEPA, 1976c, 1991).

2.25.3 The Model HSPF

The HSPF model has been successfully used in many large watershed-scale investigations that incorporate rainfall-runoff flow and contaminant loading concepts. HSPF is a 1-dimensional transient analysis, steady flow model employing kinematic wave routing techniques (e.g. uniform flow via Manning's equation). Precipitation is the basic forcing function of the model. It has the capability to model both flow and transport and fate of sediments/contaminants in both surface and subsurface environments, on a time sequenced, compartmentalized basis.

HSPF is a relatively complex "USEPA-operational" validated model that has been successfully applied to sediment and contaminant loading problems on a watershed-scale for the past twenty years with ongoing development and improvement. The primary reason HSPF serves as a model of interest in this discussion is due to its generally robust applicability and its specific capabilities to handle complex fate and transport of specific and generalized contaminants. HSPF has also been successfully linked to GIS applications (Donigian *et al*, 1995; USEPA, 1981).

These integrative approaches to NPS and PS modeling show promise for ecological risk assessment applications, as does the continued integration and automation of GIS processing. HSPF is for example the key watershed simulation routine in USEPA's BASINS program that links to the GIS package ARCView. This program will likely receive widespread use over the next ten years in the administration of individual state requirements to develop multiple watershed management plans, 10 per year for each state. For example bids were announced in May 1999 for Virginia's second round of watershed modeling projects associated with Section 319 of the Clean Water Quality Act (e.g. in this case for assessment of fecal coliforms).

Regarding integration of complex aspects of NPS management, the following excerpts are instructional (Bicknell *et al*, 1993; McKeon *et al*, 1988; USEPA, 1991):

> *"[HSPF] is the only comprehensive model of watershed hydrology and water quality that allows the integrated simulation of land and soil contaminant run-off processes with instream hydraulic, water temperature, sediment transport, nutrient, and sediment-chemical interactions"*

> *"The water quality routines for sediment erosion, pollutant interaction and groundwater quality are superior in HSPF, and the capabilities to efficiently handle <u>all</u> types of land uses and pollutant sources (including urban and agricultural, point and non-point), is a definite advantage when needed for large complex basins."*

HSPF, in its current form though is still severely limited in its ability to handle stream sediment transport mechanics of gravel bed reaches, including address of concepts of pavement, subpavement, and near equal mobility (Diplas *et al*, 1992b). Thus there are weaknesses that inherently limit the applicability of even the best scenarios conceived for NPS management and hydrologic simulation.

2.25.3.1 Flexibility and Unique Advantages of HSPF

HSPF is better set up to handle an integrated routing scheme for subbasins of the watershed than is SWRRB. The uses of HSPF are wide ranging and include recent code-improvements of interest to Leading Creek's acid mine drainage problem (Donigian *et al*, 1995):

- Flood Control and Planning Operations
- Hydropower studies

- River Basin and Watershed Planning
- Acid Mine Drainage Studies
- Storm Drainage Analysis
- Water Quality Planning and Management
- Point and Non-Point Source Pollution Analyses
- Soil Erosion and sediment transport studies
- Evaluation of urban and agricultural best management practices (BMPs)
- Fate, transport, exposure assessment, and control of pesticides, nutrients, and toxic substances
- Time-series data storage, analysis, and display.

The USEPA module ACIDPH was added to the HSPF-Version 10 to allow simulation of pH in aluminum and carbonate dominant equilibria, addressing alkalinity and iron complexation and iron competition with aluminum (Donigian *et al*, 1995). The ACIDPH model was developed after streams in the Pennsylvania Coal Mining Region, similar to Leading Creek in many ways. The ACIDPH model simulates the following species: total aluminum, free Al^{3+}, total iron, and free Fe^{3+}, H^{+}, total inorganic carbon, and alkalinity, where the HSPF-CONS module can be used to track sulfate and fluoride. The ACIDPH module has only had very limited testing thus far (Donigian *et al*, 1995), but could be ideally suited for Leading Creek's water quality problems.

2.25.4 Monte Carlo Simulation

A simulation model can also be constructed without use of direct time sequence data files. An example of this type of statistical time simulation model would be a Monte Carlo analysis. For example, looking at a cadmium-laced wastewater discharge, one could construct a Monte Carlo analysis from the daily flow distributions of the stream and the effluent, using also the stream-background and pipe concentration distributions. One could in this case simulate a random sample of values from the population of expected downstream concentration values, after mixing. In this type of stochastic simulation, caution is advised of course when any inter-dependence may exist between distributions; like flow rate and concentration of the industrial effluent. Monte Carlo can also be used more directly as a statistical test regime for evaluating various aspects of uncertainty in hydrologic simulation and other watershed modeling approaches.

2.25.5 Model Calibration, Validation, and Verification

Model calibration represents the initial and final tuning of a model, developing parameter specifications, parameter estimates and input data sets (Sorooshian *et al*, 1995). In the case of weighted-index scoring systems, the model weights or parameter estimates are often pre-defined. Model verification usually includes the practice of testing the predictability of the model with an external data set, not used in the model calibration process. Thus any model's ability to predict output accurately can be tested by collecting an independent set of input-data and output data and evaluating the model with that data. In the cases of multiple linear regression analysis (MLRA) and simulation models, output data may also be needed to verify the model's ability to predict the outcomes of interest.

Model validation implies a condition as to whether or not the model being used solves the underlying equations it represents (McKeon *et al*, 1988; USEPA, 1991). In the case of weighted-index scoring schemes (Cherry *et al*, 1999), testing validity is not often feasible. Such models are often simply defined and are assumed to be valid if the summation equation is properly executed. Weighted-index scoring schemes rely heavily on subjective professional judgement of creators and users. In the case of a MLRA application to assess various interconnections between landscape and ecological data, for example, the model can only be said to explain the underlying data set. Validity also in this case would not necessarily imply cause and effect. In the case of simulation models HSPF and SWRRB, several test cases have been performed which indicate the models themselves have been validated and behave as expected under various combinations of inputs and boundary conditions (Arnold *et al*, 1995; Donigian *et al*, 1995). In some simulations, a physical basis exists and cause and effect are inherently developed in the modeling process.

2.25.6 Uncertainty Analysis

Beck (1987) originally described four areas of uncertainty for water quality modeling. These related to:

(1) Model structure,
(2) Model parameters,
(3) Aggregate uncertainty in future estimates, and
(4) Data used for reduction of critical areas of model uncertainty.

Each of the above aspects is obviously also important to simulation model outcomes. Descriptions of uncertainty in hydrologic simulations will be overlapped by evaluation of four key areas described by Melching (1995):

(a) Natural randomness,
(b) Input data,
(c) Model parameters, and
(d) Model structure.

The discussion of uncertainty can be constructed in a framework of reliability in model output, applied to both estimation and forecasting (Beck, 1987; Melching, 1995). Melching indicates uncertainty information required by various reliability estimation techniques will typically include data on the mean, variance, and in some cases the probability distribution functions (PDF's) or cumulative distribution functions (CDFs) of basic variables describing the watershed. From this data various model estimators are derived. Melching also gives a general indication that for Monte Carlo Simulation (MCS) methods, the correlation between the parameter values and the model output values are assessed, where those with high correlation are important (Melching, 1995).

Uncertainty analysis associated with parameter estimation can be discussed as being either subjective or objective. In the objective case, linear and non-linear measurement functions are constructed to measure uncertainty (e.g. MAD, OLS→BLUE). Sensitivity, residual, regression, and leverage analysis can be applied to water quality models for estimating uncertainty. Established by several authors, sensitivity analysis, evaluating the Jacobian matrix (state variable rates of change) is not usually amenable to conducting uncertainty analysis in watershed simulation models (Melching, 1995). Sensitivity analysis does not always identify model uncertainties that effect model outcomes. It has been determined that traditional sensitivity analysis, a standard technique of perturbing one variable at a time, is not typically appropriate for conducting uncertainty analysis in watershed simulation models (Melching, 1995). Lin and others report that application of the stochastic integral equation method has been shown to be capable of including both model and input uncertainties into the rainfall-runoff model (Lin *et al*, 1995).

MCS methods remain the benchmark technique against which other reliability measurement techniques will be compared (Beck, 1987; Melching, 1995). Some uncertainty analysis techniques build on others like the Generalized Likelihood Uncertainty Estimation that uses MCS. Other techniques for conducting uncertainty analysis in hydrologic simulation might include Latin Hypercube Simulation, Mean-Value First-Order Second-Moment, Advanced First-Order Second-Moment, Rosenblueth's Point Estimation Method, Regional Sensitivity Analysis, and Tree Structured Density Estimation. Regional Sensitivity Analysis for example identifies basic-key model variables, determined with a simulation procedure that segregates output data assuming a hypothesized range of reasonable values (Melching, 1995).

2.25.7 Advantages/Disadvantages of Simulation Models

The main advantage of simulation models is that once set up they can be used to quickly and efficiently test various management schemes (e.g. reduced loading scenarios). This can be done quantitatively. The simulation models are readily able to evaluate landscape-level impacts upon the quality and quantity of water (Bedient *et al*, 1992, Linsley *et al*, 1975). A major advantage of simulation models is the insight gathered from the shear detail and labor that goes into compiling such a complex model application (Bedient *et al*, 1992). This implies that one will gain a fair amount of insight into the system simply by reviewing the watershed in this manner. The advantages awarded to extensive data review, both graphical and analytical, can also be recognized with statistical model tools, and to some degree with biological or ecotoxicological weighted index approaches (Cherry *et al*, 1999; OEPA, 1994; USEPA, 1996c).

Drawbacks to simulation models include the amount of data needed to achieve valid, calibrated results, where many applications of simulation models may not be properly calibrated and verified. Model accuracy is largely determined by model input (Bedient *et al*, 1992; Bicknell *et al*, 1993; Donigian *et al*, 1995; Linsley *et al*, 1975). HSPF and SWRRB flow models can also be calibrated with stage-discharge measurements available often available in watershed monitoring programs or through the USGS. Sediment run-off and pH modeling would need additional calibration data. With adequate input data, the simulation models would be expected to have good predictability. The models can then be verified with a data set not used in the calibration of the model (McKeon *et al*, 1988; Sorooshian *et al*, 1995).

2.26 MULTIVARIATE REGRESSION AND OTHER STOCHASTIC APPROACHES

The following information outlines the uses of statistical regression models and how a multiple linear regression analysis (MLRA) could be constructed for Leading Creek. Standard texts on statistics cover the area of regression analysis in detail (Rencher, 1995; Zar, 1999).

With respect to basic applications of multiple regression, excerpts from the NCSS 97 users manual are given below with materials originally taken from Montgomery (Hintze, 1997). The outline summarizes when and how regression analysis might be used in modeling various systems:

Coefficient Estimation - *The analyst may have a theoretical relationship in mind, and the regression analysis will confirm this theory. Most likely, there is specific interest in the magnitudes and signs of the coefficients. Frequently, this purpose for regression overlaps with others.*

Prediction - *The prime concern here is to predict some response variable. These predictions may be very crucial in planning, monitoring, or evaluating some process or system. There are many assumptions and qualifications. You must not extrapolate beyond the range of the data.*

Control - *Regression models may be used for monitoring and controlling a system. For control purposes, the independent variables must be related to the dependent variable in a causal way.*

Variable Selection or Screening - *In this case, a search is conducted for those independent variables that explain a significant amount of the variation in the dependent variable. In most applications, this is not a one-time process but a continual model-building process.*

Principal Components Analysis (PCA) is an analytical technique. While fundamentally different from regression analysis, PCA can also be used to identify an optimum set of variables. PCA can for example be combined with standard regression approaches to identify candidate reduced order models (Rencher, 1995; Zar, 1999; Hintze, 1997). For Leading Creek, all of the above regression purposes might conceivably be folded into the modeling process. A PCA could for example be run on the Leading Creek watershed variable sets to evaluate or determine a regressor set for Total Taxa or another measure of biodiversity (Rencher, 1995).

On a tangent note, controllability and observability of variables in large-scale physical (state variable) systems has also been studied in other disciplines such as control systems in electrical engineering (Lindner *et al*, 1988, 1989). In control feedback systems, model reduction is often pursued along somewhat analogous lines. This is done by seeking-out (and then removing) variables in a state space model that are over parameterized or are otherwise either uncontrollable or unobservable (or nearly uncontrollable or nearly unobservable) (Babendreier, 1987; Lindner *et al*, 1989).

2.26.1 Multiple Linear Regression Analysis - MLRA

Multiple Linear Regression Analysis (MLRA) is a useful tool for variable reduction and data screening (Rencher, 1995). MLRA is a statistical approach often based on the familiar least square solution or some other suitable control metric. The reduced order model is directly related to a smaller set of the original variables via a linear combination of regressor or independent variables.

MLRA is a quantitative procedure offering a universally accepted stochastic approach to multiple variable analysis. Conceptually, quantitative (stochastic) evaluation of available data could be used for example to reduce the large Leading Creek database collected (Gallagher *et al*, 1999). The product of the MLRA will often be a reduced order model distilling the database from many variables down to a few key variables. This process, if supported by knowledge and separate lines of evidence, can potentially offer an optimum set of environmental parameters to gage biodiversity trends (USEPA, 1996c). This optimum set of variables could be used to hopefully support final decisions and remedy selections in a stream restoration effort (USEPA, 1996c, 1998; USDA, 1998).

2.26.1.1 Advantages of Statistical Modeling Approaches

Statistical based models such as MLRA represent modeling processes that incorporate certain statistical criteria (Rencher, 1995; Zar, 1999). Criteria such as using R^2 as a measure of fit or requiring a certain level of statistical significance of the linear regression can be used to ensure that the final model selected meets acceptable standards. This could also include for example testing collinearity of regressors.

A distinct advantage of the statistical approach over weighted-index scoring systems (Cherry *et al*, 1999) is that the MLRA retains a desired set of statistical properties. The model retains a quantifiable expression of significance, inherently determined by the modeling effort. Weighted-index scoring systems on the other hand have no similar underlying statistical significance. The MLRA model also statistically relates the relative % of variation of a key dependent biodiversity variable to meaningful parameters that can be influenced through improvement projects. Another feature of the MLRA is that the approach represents a unique algorithm that operates, for example, on R^2 as a measure of the best model. The benefit of an MLRA approach is that it can further support, in a comprehensive systematic fashion, analytical, statistical, and theoretical monitoring and analysis approaches undertaken during data screening.

2.26.1.2 MLRA Applications in Ecological Risk Assessment

As an alternative approach to the multimetric approaches such as IBI and ICI for assessing biotic integrity, one can directly model some dependent variable(s) of choice as a function of watershed factors. Logically these factors would include those aspects of an available data set that can best relate or capture the desired property of watershed and subwatershed biotic integrity or biodiversity. Historical approaches in this area have generally focused on physical and chemical factors (Moss *et al*, 1987; Wright *et al*, 1984).

There are generally two approaches taken in the application of multivariate approaches for ecological risk assessments. In the first approach, a classification of abundance data is first determined for available reference sites. This can be done using clustering or ordination techniques (Ludwig *et al*, 1988). With the class assignments determined, discriminant analysis is conducted to associate membership criteria for physical-chemical data available (Wright *et al*, 1984). Subsequently, a test site is evaluated using the discriminant functions developed (Moss *et al*, 1987; USEPA, 1996c).

The alternate to discriminant analysis prescribed by USEPA is direct analysis between species composition variables and environmental variables using methods such as canonical correlation analysis, canonical correspondence analysis, or multidimensional scaling. USEPA also points out that multivariate approaches for bioassessment are still under development, but nonetheless, as this approach becomes more refined may prove to be a viable option (USEPA, 1996c). One major issue underscored at this time is that assessment thresholds and standard procedures for multivariate assessments are not yet well developed, aside from the application of professional judgement (USEPA, 1996c).

Having only one or a few sampling events for all variables may often be the case in multidisciplinary ecological risk assessments. Complex assessments can still employ multivariate regression analysis to reduce sparse event/location databases where # final regressor variables < (# sites monitored) * (# sampling events) < # total system variables. In these cases a structured multivariate analysis broken down by various data groups (or disciplines) must first be applied in some manner, imparting some aspect of outside knowledge of the system of variables. In addition to chemical related phenomena, a multivariate regression analysis describing aquatic biodiversity in a watershed may also identify key physical factors relating the importance of habitat condition (USEPA, 1996c).

From the available research in this area, methodical approaches in watershed-scale ecological risk assessment are further needed in handling data screening, data analysis, and modeling of the integral physical-chemical-biological ecosystem.

2.26.2 Statistical Methods and Calculations

For initial data screening, bivariate correlation and graphical analysis between various investigative discipline data sets can be used to evaluate which watershed system parameters trend with metrics of biodiversity or which also strongly relate to other external endpoint group variables. Example endpoint groups are described below. Principal components analysis, discriminant analysis, canonical analysis, regression analysis, and cluster analysis, along with possibly other pattern recognition techniques may be used to evaluate the effects of water chemistry, sediment chemistry, and sedimentation upon biodiversity (Breen *et al*, 1985; Brereton, 1990; Burton *et al*, 1992; Johnson, 1998; Ludwig *et al*, 1988; Moss *et al*, 1987; Rencher, 1995; Richards *et al*, 1994b; Shin, 1982; USEPA, 1992, 1996c; Wright *et al*, 1984; Zar, 1999). Specific attributes and applications of these techniques are discussed in detail below.

2.26.2.1 Endpoint Group Organization

In facilitating both bivariate and multivariate analysis, task areas of site monitoring, data and variable management can be broken down and normalized into various endpoint groups for statistical analysis and comparison. The variable data groupings represent a functional structure of the underlying database that can initially be used to organize a watersheds information system.

Each watershed investigation will need to define a unique approach depending upon the final monitoring system design utilized. An example framework for Leading Creek is described by:

- Biodiversity
- Biomass
- RBP habitat
- RIS habitat
- Sediment depth
- Sediment particle size
- Sediment toxicity
- Sediment chemistry
- *In situ* clam toxicity
- Acute toxicity
- Channel geometry and hydraulics
- Land use/cover
- Water column concentration
- Water column load rates
- Water column unit area load rates

2.26.2.2 Univariate-Bivariate Data Screening Approaches

The role of quantitative measures of hydrologic and hydraulic watershed properties and their impacts on ecological risk assessment can be investigated through use of an integrated GIS database system as developed here. As a first pass through various endpoint groups, Pearson correlation coefficients or Spearman-rank correlation coefficients can be examined between metrics of biology, ecotoxicology, and physical-based parameters describing channel, velocity, and flow condition at each test site. Because of its role and dominant influence, many of the graphics tools that can be developed can also include a grouping concept that can separates sites based on the %area of the subwatershed covered, for example looking at surface mines. Land use/cover data, due its geomorphologic relationship with flow (Leopold *et al*, 1964), is best utilized by normalization to unit drainage area.

For univariate and bivariate data screening in Windows-based personal computer environments, the Leading Creek database created was capable of querying up to 254 system variables simultaneously filtering, analyzing, ranking, sorting, and graphing up to 32,258 (or more) unique x-y relationships among those variables (Babendreier *et al*, 1998b, 1998c). In complex multidisciplinary risk assessments, only a small fraction of system-wide x-y relationships will be significantly correlated. It is anticipated some system representing this capability can be developed and used as a powerful filter for conducting bivariate analysis of any watershed data set. This tool and approach can allow identification of significant system trends variables relating to ecological concerns. Graphically, non-linear trends can also be similarly viewed in some cases via transformations readily available. Log-linear and exponential relationships are examples of data screening can be extended to account for this functionality.

The advantage provided by a spreadsheet controlled data screening approach suggested is that in a multi-metric project covering many disciplines, the number of pairs of data for any two variables can be concurrently addressed. Across all variable combinations the range can vary significantly (e.g. ranging from 2 pairs upwards). Thus for example if one were to rank all R^2 values for 254 variables, it may become critical to determine if one relation is based on two data points or twelve, and the other on 30 points. This last type of scenario, unbalanced sample design, is likely to occur in assessments that build upon one study to the next for a given system. Simple computational tools as such along with others that can be used are not new or novel. Their application in ecological risk assessment is though sometimes greatly inhibited by the complexity of integrating the many areas investigated (or endpoint groups of data analyzed).

A well-defined data screening process can be instrumental in eventually discerning watershed scale relationships that join multiple approaches (e.g. GIS, database, spreadsheet, statistical, and graphical linkages for data screening). When variables from discipline to discipline are not readily normalized, the task of interdisciplinary data screening can be made more difficult and can render meaningless or misleading relationships. This represents perhaps the single greatest challenge in integrating multidisciplinary approaches to ecological risk assessment.

A useful screening system for large variable sets would seem then to be in need of at least the following attributes and abilities:

- ❖ Ability to scan many variables together at once, as many as feasibly possible.

- ❖ Ability to concurrently calculate and rank a useful estimator of all possible relations for a given sub-selection of all system variables (e.g. a linear correlation R^2 coefficient for instance between each variable and all others, and/or calculated upon transformed variables, etc.).

- ❖ Ability to maintain easy filter control over the number of pairs of data that were used to estimate the strength of any given relation or ranked variable set.

- ❖ Ability to include or exclude various endpoints within and between endpoint groups for a given analytical session (e.g. comparing two key biodiversity metrics to all chemistry concentration parameters, looking at either averages or maximum values)

Filter control over other attributes of variables to chose from would also be highly desirable in such a screening system, where two groups of variables could be quickly separated and zeroed in on for special analysis in answering questions posed by various team researchers. With such a filter for example, growth rate for each clam test event could be quickly selected and viewed against drainage area or 10 separate categories of land use/cover. These types of strategies might become more useful for instance after initial ranking efforts of a system-wide database have been completed. The examples are illustrative and impart the potential utility of a well-organized data screening approach for large-scale, multivariate investigations that can conceivably evaluate hundreds to thousands of directly monitored or constructed variables. Data screening is an invaluable step in risk assessment data analysis and should always be applied prior to use of more formal multivariate techniques.

2.26.3 Multivariate Data Analysis Approach

The Leading Creek system is over-parameterized for statistical data analysis, which may often be the case in large-scale investigations of stream ecology. To evaluate and further resolve the inherent modeling dilemma, a variety of techniques to analyze and reduce data can be employed and results effectively compared. The various data analysis techniques introduced here can be applied to a watershed system database which might for example represent 500 variables, where only 25 to 50 data points may be available across all variables simultaneously. To complete system-wide data analysis, various analytical and statistical techniques can be used to evaluate the Leading Creek database. These can first be logically broken down by various investigative disciplines given by endpoint groups described above. For ecological risk assessments, as previously discussed in paradigms for multimedia and multidisciplinary risk assessment, this may be focused with respect to discerning trends in biodiversity metrics. During such an approach, each endpoint group though may at some point take on an interim focus as a set of dependent variables. For example, watershed land use/cover might be analyzed with respect to water quality data sets.

System-wide data analysis approaches might include some form of principal component analysis (PCA) application, multi-variable/variate regression analysis (MLRA), canonical correlation analysis (CCA), canonical variate analysis (CVA), canonical discriminant analysis (CDA), and potentially cluster analysis (Johnson, 1998; Rencher, 1995). Note canonical variate analysis is sometimes also referred to as multiple discriminant analysis in ecological writings (Shin, 1982; Wright *et al*, 1984), which is different from (discrete) discriminant analysis, the latter is also sometimes referred to as classification analysis (Johnson, 1998).

2.26.3.1 Comparing Multivariate Methods

In addition to multivariate regression analysis techniques, PCA (and factor analysis), CDA, CVA, and CCA are all key techniques for reducing model dimensionality (Johnson, 1998, Rencher, 1995). PCA is also commonly used as a data-screening tool. On the other hand, the primary role of discriminant analysis (DA) is to predict the population from which an observation is most likely to have come from (Johnson, 1998). Historically, canonical variate analysis (CVA) along with PCA, pattern recognition, cluster analysis, and correspondence analysis have been popular in ecological statistics (Burton, 1992; Ludwig *et al*, 1988; Moss *et al*, 1987; Shin, 1982; Wright *et al*, 1984), and chemometrics (Breen *et al*, 1985; Brereton, 1990, 1992). Correspondence analysis is a discrete analogy for PCA that uses a Chi^2-value.

Canonical correlation analysis is the study of the linear relations between two sets of variables. It is the multivariate extension of correlation analysis (Johnson, 1998). Discriminant analysis, MANOVA, and multiple regression are all special cases of canonical correlation (Rencher, 1995). According to the NCSS Software Users manual, CCA provides the most general multivariate framework (Hintze, 1997). Because of its generality though, CCA is the often least used of the multivariate procedures.

Canonical discriminant analysis (CDA) is a procedure that creates new variables that contain all of the useful information available for discrimination in the original variables

(Johnson, 1998). CDA is also sometimes referred to as Fisher's "between-within" method, and is somewhat analogous to PCA but is computed differently (Johnson, 1998; Rencher, 1995). Canonical variate analysis (CVA) is yet another approach that creates new variables in conjunction with multivariate analysis of variance. CVA in essence allows one to compare means of m variables in a reduced canonical space of dimension k, where k < m, in a way allowing one to compare the relative distances of all means, by using all of the measured variables simultaneously (Hintze, 1997).

Discriminant analysis finds a set of prediction equations based on independent variables that are used to classify individuals into groups. According to the NCSS User's Manual, there are two possible objectives in a discriminant analysis: (1) finding a predictive equation for classifying new individuals or (2) interpreting the predictive equation to better understand the relationships that may exist among the variables (Hintze, 1997). Discriminant analysis parallels multiple regression analysis in several ways though regression analysis deals with a continuous dependent variable, while discriminant analysis must have a discrete dependent variable (Hintze, 1997). Cluster analysis is in a sense a counterpart of discriminant analysis but deals with classification problems when it is not known before hand which subgroups that observations originate from. In discriminant analysis the difference would be that some random samples have been obtained beforehand for each unique subgroup defined (Johnson, 1998; Rencher, 1995).

All of the multivariate techniques are similar in some way and can share common attributes or elements in their approach to model reduction (e.g. reducing the number of variables needed to successfully describe the system or dependent variable set). For example, functions can be developed relating attributes of multivariate regression analysis with principal components and canonical analysis (Rencher, 1995). CCA is best noted for its ability to compare groups of variables, where PCA (and factor analysis) can also sometimes be used in this way. In general, PCA, CDA, CVA, and CCA create new variables, where discriminant analysis (DA) doesn't (Johnson, 1998).

2.26.3.2 Multivariate Analysis Application on Leading Creek

A given approach for analysis might investigate use of Principal Components Analysis (PCA) to identify potentially dominant variables from each investigative discipline (or endpoint group). PCA might be used in this way to evaluate all endpoint data groups described for example, along with their internal relationships, to first derive a reduced set of regressors for later use in MLRA. Secondly, PCA and MLRA may also be used more directly to analyze all variables from all groups in context of their relation to biodiversity metrics.

The other forms of multivariate statistical analysis might then also be investigated including canonical correlation and canonical variate (multiple discriminant) analysis. CCA will primarily be used to evaluate groupings of variables that may be able to further assist in objective model reduction. In this way, multivariate (primarily multivariable) regression analysis might be attempted on various selected subsets of watershed variables informed by PCA, CDA, CVA, and CCA analysis and other data screening and modeling results. Multivariate regression analysis in the watershed database system described for Leading Creek is likely more useful than discrete discriminant analysis (e.g. classification analysis).

The purpose of a system-wide watershed data analysis effort might be to identify dominant anthropogenic stresses and other natural physical and chemical variables that determine overall aquatic biodiversity. Ideally, a predictive equation would be sought that can relate the key dominant system variables (for example pH, water quality, and sediment size) to some useful representative metric of overall system biodiversity, such as Total Taxa. More direct, a similar function might be established directly between some land use variable and biodiversity (e.g. %AUML, %ASML, or %Bare Soil).

2.26.4 Multivariate Analysis Examples

For descriptive purposes, brief example applications of PCA, MRA, and CCA multivariate techniques are given. This provides an indication together with information above of the relative differences among most of the techniques discussed. All of the following definition statements and example data sets are taken directly from the NCSS tutorial and User's Manual (Hintze, 1997)

2.26.4.1 Principal Components Analysis (PCA) (Adapted from Hintze, 1997)

PCA is a data analysis tool that is usually used to reduce the dimensionality (number of variables) of a large number of interrelated variables, while retaining as much of the information (variation) as possible. PCA calculates a non-correlated set of variables (i.e. referred to as factors or principal components). These factors are ordered so that the first few retain most of the variation present in all of the original variables. Unlike its cousin technique Factor Analysis, PCA always yields the same solution from the same data (apart from arbitrary differences in the sign). PCA is a data analytical rather than statistical method, resolving to an eigenvalue-eigenvector problem statement (Hintze, 1997; Rencher, 1995). An example input set and results are given in Table 40 (Hintze, 1997).

Table 2.40 – Principal Components Analysis Example Data Set
(Adapted from Hintze, 1997)

Col. x1	Col. x2	Col. x3	Col. x4	Col. x5	Col. x6
50	102	103	70	75	102
4	2	5	11	11	5
81	98	94	5	85	97
31	81	86	46	50	74
65	50	51	60	57	53
22	30	39	17	15	17
36	33	39	29	27	25
31	91	96	50	56	85

Rows two and three of the data set were modified in this example to be outliers so that their influence on the analysis could be observed. As Hintze points out in this example, even though these two rows are outliers, their values on each of the individual variables are not outliers. The example shows one of the challenges of multivariate analysis where multivariate outliers are not necessarily univariate outliers.

In other words, a point may be an outlier in a multivariate space, and yet one cannot detect it by scanning the data one variable at a time. In this example, the following eigenvalue scree plots in Table 41 can be used to evaluate how many factors might be retained in a reduced order model for example. A common rule-of-thumb is to retain those factors whose eigenvalues are near to or greater than one. In this example one would retain the first two factors, noting each of the input variables used in the example was actually a weighted linear combination of two original variables.

Table 2.41 – Principal Components Analysis Example Results
(Adapted from Hintze, 1997)

Factor #	Eigenvalue	Individual Percent (%)	Cumulative Percent (%)	Scree Plot
1	4.56263	76.04	76.04	|||||||||||||||
2	1.17151	19.53	95.57	||||
3	0.24283	4.05	99.62	|
4	0.02288	0.38	100	|
5	0.00011	0.00	100	|
6	0.00004	0.00	100	|

2.26.4.2 Multivariate Regression Analysis (Adapted from Hintze, 1997)

Multivariate Regression Analysis (MRA) or more specifically multiple linear regression analysis (MLRA) refers to a group of techniques for studying the straight-line relationships among two or more variables. Although the regression problem may be solved by a number of techniques, the most-used method is least squares. A NCSS example tutorial problem is given below in Table 42. The data come from a study of the relationship of several variables with a person's I.Q.. Fifteen people were studied in the NCSS example where each person's IQ was recorded along with scores on five different personality tests.

Table 2.42 – Multivariate Regression Analysis Example Data Set
(Adapted from Hintze, 1997)

Row	Test #1	Test #2	Test #3	Test #4	Test #5	IQ Result
1	83	34	65	63	64	106
2	73	19	73	48	82	92
3	54	81	82	65	73	102
4	96	72	91	88	94	121
5	84	53	72	68	82	102
6	86	72	63	79	57	105
7	76	62	64	69	64	97
8	54	49	43	52	84	92
9	37	43	92	39	72	94
10	42	54	96	48	83	112
11	71	63	52	69	42	130
12	63	74	74	71	91	115
13	69	81	82	75	54	98
14	81	89	64	85	62	96
15	50	75	72	64	45	103

A typical output is given below in Table 43 for the MLRA technique. The results give the predictive coefficients of the linear equation relating the independent variables to the dependent variable(s) chosen, including an intercept value. Features of the multiple regression approach include a null hypothesis test evaluating the relationship of each variable in the equation to the regression, along with an overall R^2 score, in this case 0.399, indicating the amount of variation of the dependent variable explained by the independent variables used. Decision level significance is an important attribute of each null hypothesis test as is the power of the ability to reject the null-hypothesis when it is actually false. Shown in the regression results in Table 43, the model in this case does not indicate a strong multivariate relationship, where added effort to examine other output would determine severe multi-collinearity among variables. Like any model, a few tests on assumptions should usually be carried out to determine the strength of the model developed (Hintze, 1997). Data screening and evaluation of outliers are among two of the more important elements of data processing prior to multivariate model applications discussed here (Johnson, 1998; Rencher, 1995).

Table 2.43 – Multivariate Regression Analysis Example Results
(Adapted from Hintze, 1997)

Independent Variable	**Regression Coefficient**	**Standard Error**	**T-Value (Ho: B=0)**	**Prob. Level**	**Decision (5%)**	**Power (5%)**
Intercept	85.24	23.70	3.60	0.006	Reject Ho	0.89
Test 1	-1.93	1.03	-1.88	0.093	Accept Ho	0.39
Test 2	-1.66	0.87	-1.90	0.090	Accept Ho	0.40
Test 3	0.10	0.22	0.48	0.645	Accept Ho	0.07
Test 4	3.78	1.83	2.06	0.070	Accept Ho	0.45
Test 5	-0.04	0.20	-0.20	0.845	Accept Ho	0.05
R-Squared	0.399					

2.26.4.3 Canonical Correlation Analysis (Adapted from Hintze, 1997)

In one example of canonical correlation analysis presented by Hintze (1997), suppose you have given a group of students two tests of ten questions each and you must determine the overall correlation between the tests. Canonical correlation finds a weighted average of the questions from the first test and correlates this with a weighted average of the questions from the second test. The weights are constructed to maximize the correlation between these two averages. This correlation is called the first canonical correlation coefficient. One can create another set of weighted averages unrelated to the first and also calculate their correlation. This correlation is the second canonical correlation coefficient. This process continues until the number of canonical correlations equals the number of variables in the smallest group.

Canonical correlation terminology makes an important distinction between the words variables and variates. The term "variable" is reserved for referring to the original variables being analyzed. The term variate is used to refer to a variable that is constructed as a weighted average of the original variables (Hintze, 1997; Rencher, 1995). Another example, the same one used above in the regression discussion, is also repeated here except here NCSS correlated variables Test 1, Test 2, and Test 3 with variables Test 4, Test 5, and IQ. Example input is found in Table 42, and example output for canonical correlation analysis is given in Tables 44 and 45.

Table 2.44 – Canonical Correlation Analysis Example Results - Variates
(Adapted from Hintze, 1997)

Variate #	Canonical Correlation	R-Squared	F-Value	Num DF	Den DF	Prob. Level	Wilks' Lambda
1	0.9956	0.99	16.58	9	22	0	0.006819
2	0.4675	0.22	0.67	4	20	0.618	0.7765
3	0.07981	0.0064	0.07	1	11	0.795	0.9936

Table 2.45 – Canonical Correlation Analysis Results - Variation Explained
(Adapted from Hintze, 1997)

Canonical Variate #	Variation In these Variables	Explained by these Variates	Individual Percent (%) Explained	Cumulative Percent (%) Explained	Canonical Correlation Squared
1	Y	Y	37.6	37.6	0.9912
2	Y	Y	32.1	69.7	0.2185
3	Y	Y	30.3	100	0.0064
1	Y	X	37.2	37.2	0.9912
2	Y	X	7	44.3	0.2185
3	Y	X	0.2	44.5	0.0064
1	X	Y	37.1	37.1	0.9912
2	X	Y	5.4	42.5	0.2185
3	X	Y	0.2	42.8	0.0064
1	X	X	37.4	37.4	0.9912
2	X	X	24.8	62.2	0.2185
3	X	X	37.8	100	0.0064

In Table 45, each row of the report presents the results of how well a set of variables is explained by a particular canonical variate. The table displays the percentage of the variation in each set of variables explained by other sets. With this and other reporting forms, the analyst can evaluate various indications of the statistical test output and examine the suitability of new variables described. As seen above and in Tables 46, 57, and 58, this test can be useful for examining relationships between groups of variables and the new variates developed. The variate descriptions from the NCSS example of test scores and IQ are found in Tables 49 and 50. In Table 45, Wilks' lambda statistic is interpreted just the opposite of R-Squared where a value near zero indicates high correlation while a value near one indicates low correlation. Probability level associated with the F-statistic is expected close to zero for strong variates, a cut-off of 0.01 to 0.05 is often used (Hintze, 1997). The F-value tests whether this canonical correlation and those following are zero (e.g. the F-value for variate 2 represents the test on variates 2 and 3).

Table 51 shows the correlations between the variables and the variates. As indicated by Hintze (1997), by determining which variables are highly correlated with a particular variate, you can determine its interpretation in some cases. For example, the variate Y1 is highly correlated with Test #4. Therefore variable Y1 may have the same interpretation as variable Test #4. One reason for CCA may be model reduction. In this technique the analogous approach to model reduction is to prescribe or choose the number of canonical variates you wish to represent the data set with, where the first canonical variate is always the strongest representation (Johnson, 1998).

Table 2.46 - Standardized Y Canonical Coefficients Section
(Adapted from Hintze, 1997)

Variable	Variate	Variate	Variate
	Y1	Y2	Y3
Test #4	1.021	0.1050	0.3709
Test #5	-0.006	0.9903	0.2240
IQ Result	-0.06536	0.2298	-1.050

Table 2.47 - Standardized X Canonical Coefficients Section
(Adapted from Hintze, 1997)

Variable	Variate	Variate	Variate
	X1	X2	X3
Test #1	0.6907	0.5925	0.5103
Test #2	0.6556	-0.4282	-0.6361
Test #3	-0.008941	0.9196	-0.4852

Table 2.48 Variable - Variate Correlations Section
(Adapted from Hintze, 1997)

Variable	Y1	Y2	Y3	X1	X2	X3
Test #1	0.7552	0.1448	0.04575	0.7586	0.3098	0.5732
Test #2	0.7210	-0.1479	-0.04891	0.7242	-0.3163	-0.6128
Test #3	-0.1509	0.3462	-0.05225	-0.1515	0.7405	-0.6547
Test #4	0.9981	0.01915	-0.05793	0.9937	0.00895	-0.00462
Test #5	-0.1788	0.9588	0.2209	-0.1780	0.4482	0.01763
IQ Result	0.3143	0.2113	-0.9255	0.3130	0.09876	-0.07387

2.27 LITERATURE CITED

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Chapter 3 - A GIS BASED WATERSHED RISK ASSESSMENT OF URBAN, AGRICULTURE AND COAL MINING IMPACTS UPON AQUATIC HABITAT

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(ABSTRACT)

The impacts of urban, agriculture and coal mining land uses upon aquatic habitat were investigated. The study was conducted in Leading Creek, a 388 km^2 watershed in southeastern Ohio with abandoned strip mine land (ASML), active deep underground mines, and abandoned near-surface underground mine land (AUML). Both directly developed and remotely sensed land use and land cover data were collected at a scale of 1:24000. Relationships among land use/cover and aquatic habitat were evaluated in environments with commingled mining, urban, and agricultural activities. Two qualitative habitat scoring protocols were used incorporating standards from USEPA and Ohio EPA. For the nine variable USEPA Rapid Bioassessment Protocol (RBP), 90% of system variation was distributed across 4 variables: scour/deposition, bank vegetation, pool/riffle-run/bend, and streamside cover. A Pearson correlation coefficient $R^2 = 0.87$ was determined between full and reduced order RBP models. System variation was distributed across all seven variables in the OEPA Qualitative Habit Evaluation Index (QHEI).

Of the 4 dominant RBP habitat subscores and total RBP score, only pool/riffle-run/bend and total RBP were significantly correlated with biodiversity and land use/cover variables studied, while no differences were significant for land use/cover thresholds evaluated. Two QHEI habitat subscores, substrate, pool-riffle, and total QHEI were significantly correlated with biodiversity and land use/cover. Sites with ASML>3% and Urban>5% had significantly impacted total QHEI scores while active mining and threshold agricultural activities represented relatively insignificant stresses to habitat. For ASML>3% with commingled AUML>2%, differences in habitat were attributed to strip mines and showed impacted pool-riffle subscores. Acid mine drainage (AMD) effects upon habitat were insignificant. Inundating sedimentation impacts for ASML were associated with increased sand fractions in bed sediments. Attributed in part to siltation, for Urban>5% the QHEI substrate subscore was heavily impacted. Unreclaimed strip mines and strip mines reclaimed since 1978 were evaluated separately with no discernable differences seen between respective correlations with qualitative habitat data.

Chapter 3 - A GIS BASED WATERSHED RISK ASSESSMENT OF URBAN, AGRICULTURE AND COAL MINING IMPACTS UPON AQUATIC HABITAT

3.1 INTRODUCTION

In many Appalachian watersheds, coal mining activities represent critical components of stream health assessment. This research focused on assessment of aquatic habitat and its associated ecological responses to active long-wall underground coal mining, abandoned underground (AUML), and abandoned surface contour mining activities (ASML). Reclamation status of abandoned surface contour mines (e.g. strip mines) was investigated in addition to commingled mixed urban and agricultural land uses. Normalized to drainage area, both directly developed and remotely sensed quantitative land use and land cover data were evaluated.

3.1.1 Aquatic Habitat Assessment

Karr identified habitat as one of five key internal variables that are subject to anthropogenic control and influence (Karr, *et al* 1986, 1991; USEPA, 1996). Substrate type, water depth, current velocity, and spatial and temporal complexity of physical habitat initially defined critical elements of habitat structure. Yoder and Rankin (1996) expanded upon Karr's biotic integrity framework (from Karr *et al*. 1986). Key components of habitat can include riparian vegetation, siltation, sinuosity, current, substrate, instream cover canopy, gradient, channel morphology, bank stability, and stream width/depth. Many features are contained within or measured by components of various qualitative habitat-scoring programs available for ecological risk assessment.

Several aspects of agricultural run-off have been investigated (Bicknell *et al*, 1993; Donigian *et al*, 1995; USDA, 1998; USEPA, 1991; Zison, 1980). Mining and reclamation activities are known to significantly change watershed soils and vegetation, and can significantly effect sediment load (Bonta *et al*, 1992; USDA, 1985; USEPA 1976, 1991). Spoil slides, haul roads and the mined area itself are for example three major sources of sediment made available for transport into streams (Curtis, 1973, Dickens *et al*, 1985). In many areas of eastern United States, including Virginia, West Virginia, and Ohio, urbanization, agriculture run-off, and mining impact represent two of the most influential and widespread classes of NPS pollution (Cherry, *et al* 1995; Smith *et al*, 1998; USDA, 1983, 1985, 1998; USEPA 1991; Waters, 1995).

Habitat assessment of watersheds has more recently focused on directly evaluating the physical influences of medium/fine sands, silts, and clays on community level macro-invertebrate diversity (Luedtke *et al*, 1976; Miller, 1988; Hawkins *et al*, 1990; Nelson *et al*, 1992, Richards *et al*, 1992, 1994b; USEPA, 1995). Substrate and cobble embeddedness are characterized by threshold impairment levels of fine materials <2mm. These characteristics are physically dependent upon erosional features of the watershed and instream hydraulics (Diplas, 1994; Richards *et al*, 1994a, 1994b; Shirazi *et al*, 1981). Strictly quantitative measures of physical instream habitat such as particle size distribution, porosity, permeability, and stream velocity may be necessary to distinguish between physical and chemical impairment in aquatic systems (Beebe, 1996; Belanger, 1985; Burton, 1992; DeWitt *et al*, 1988; Diplas; 1994; Richards *et al*, 1994a; Sprague, 1985; Shirazi, 1981).

Compared to any single quantitative metric, site or reach based qualitative habitat evaluations may best discern land use/cover impacts to aquatic habitat. Yoder and Rankin (1996) indicated that biological communities respond predictably to gradients of environmental impact which chemical/physical water quality criteria alone cannot adequately discriminate or sometimes even detect. Habitat degradation and sedimentation are two widespread impacts of nonpoint source origin that are difficult to measure through chemical/physical assessments alone (Yoder *et al*, 1996). Richards and Host (1994b) initiated a discussion of potential benefits that 1:250000 scale and finer geodata can have in predicting landscape-habitat relations. Finer resolution may be needed to relate land use to biometrics. Yoder (1996) further emphasized that biological assessments must be accompanied by appropriate characterizations of habitat quality and land use statistics. Both may be needed to adequately establish relationships between stressors that impact and degrade aquatic ecosystems, and resultant quality of the ecosystem.

3.1.2 Study Objectives and Approach

The hypothesis investigated here addressed hydrologic-based watershed-scale ecological risk assessment by employing Geographical Information System (GIS) and land use/cover analysis. The objective was to evaluate quantitative descriptions of agriculture, urban, surface mining, and underground coal mining, relating their impacts to qualitative and quantitative habitat. Few active and abandoned mine studies exist that combine GIS at fine scales of 1:24000, land use/cover analysis, and ecological risk assessment tasks (Richards *et al*, 1994b, 1996). This paper integrated these approaches by testing the hypothesis that, at a scale of 1:24000, land use/cover variables describing urban, agricultural, and various mining activities can be used to characterize risk to physical habitat.

The hypothesis was investigated by simultaneously comparing habitat to land use/cover and benthic macroinvertebrate biodiversity (Richards *et al*, 1994b). To evaluate land use/cover in an integrated risk assessment framework, collection and comparison of spatial and temporal data describing four profiles of a watershed were undertaken: (a) hydrology-hydraulics, (b) chemistry of natural sediments and waters, (c) insect and fish community assemblage, and (d) comparison of responses to toxicants shown for lab and *in situ* test organisms. Quantitative and qualitative instream and riparian habitat assessments were conducted (Anderson *et al*, 1976; Babendreier *et al*, 1998; Delong *et al*, 1991; Gallagher *et al*; 1999; Rankin, 1989;USEPA, 1989).

3.2 STUDY WATERSHEDS

Leading Creek is a fifth order stream (scale: 1:24000) 96% of which spans Meigs County, with other portions in Athens, and Gallia Counties. Together, 11 mainstem stations and 19 tributary stations shown in Figure 3.1 were selected for hydrologic, hydraulic, and ecological risk assessment monitoring. Mainstem stations were located to provide complete longitudinal coverage of Leading Creek, and discrimination among potential sources of sediment or contaminant impacts to the stream system. Stream stage was monitored weekly at 10 mainstem and 6 tributary stations while continuous flow and water quality sampling were conducted at two mainstem sites. Biological monitoring characterized the surrounding reach, and cross-section sampling defined collection of sediment, water quality, and hydrologic-hydraulic properties. Leading Creek falls within the Upper-Ohio Shade subbasin hydrologic cataloging unit (HUC) 05030202 (Seaber *et al*, 1987). The creek represents 10.8% by area of the HUC.

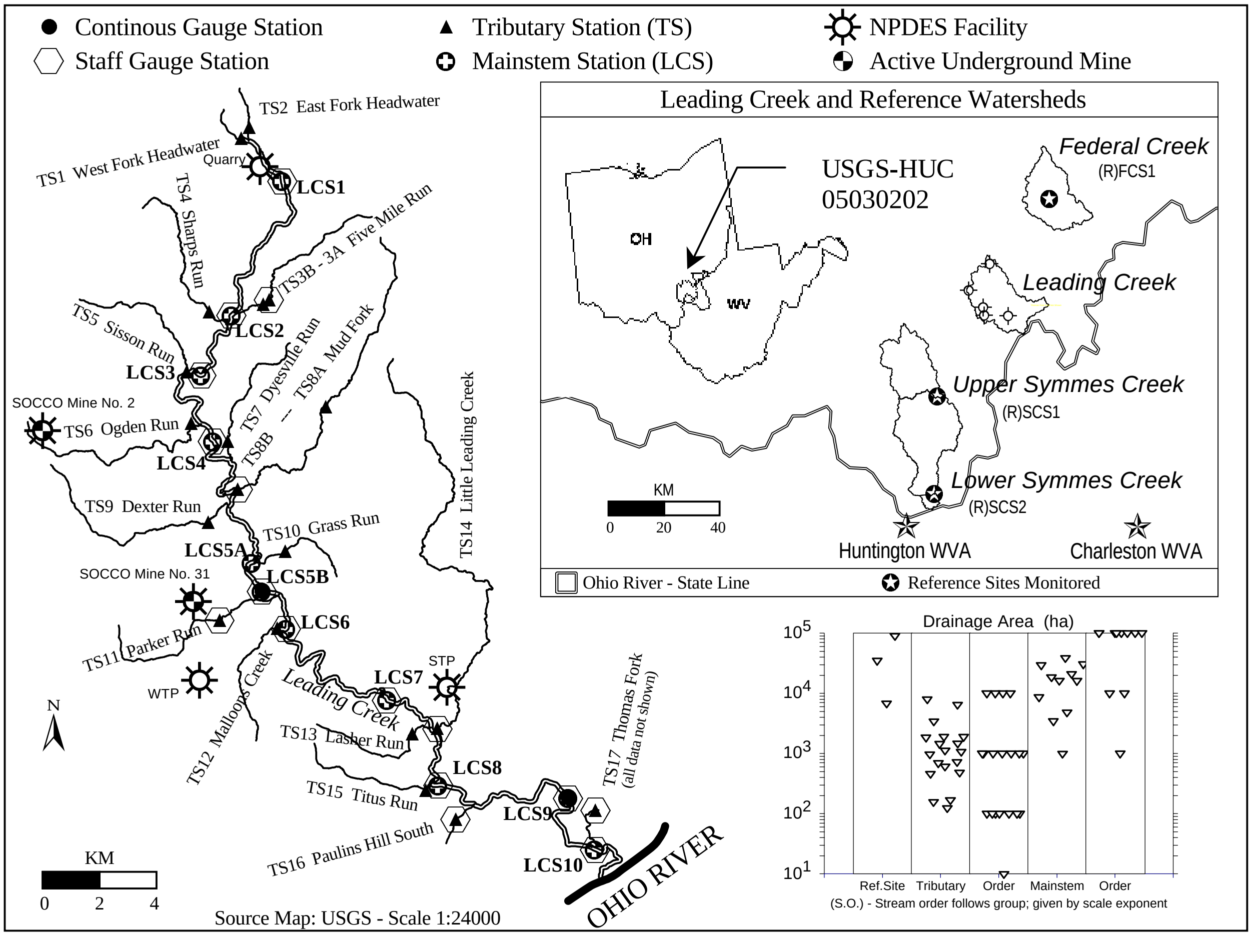


Figure 3.1 - Leading Creek Monitoring Network and Reference Watersheds

Active deep long wall mining operations are still conducted in the watershed today. Parker Run (TS11) and Ogden Run (TS6) in Figure 3.1 receive active coal mining treatment plant effluents, assorted stormwater, and minor sanitary discharges. Other NPDES sites relate to minor sanitary treatment (STP), water treatment (WTP), and quarry discharges. The upper mainstem is dominated by agriculture. Active mining and AML influence the middle to lower mainstem. AML and AMD can dominate in tributaries below LCS7 with significant, consistent AML-AMD water quality impacts on the mainstem seen below Thomas Fork.

3.2.1 Ecoregion Reference Watersheds

Monitoring included 3 reference sites outside of Leading Creek. Comparative reference stream sites in two watersheds were selected on the basis of (1) exceptionally high IBI/ICI biodiversity scores in historic assessments, or (2) visibly exceptional habitat characteristics that are representative of what might be attainable in the Western Allegheny Plateau ecoregion. IBI refers to the site's Index of Biotic Integrity, a fish-based metric, and ICI refers to the Invertebrate Community Index, an insect-based metric (Karr, 1991; Miller *et al*, 1988; OEPA, 1989; USEPA, 1996). Upon initial inspection, the reference sites did not indicate marked decline but were potentially impacted by various anthropogenic traits including adverse effects from urban, agricultural, ASML, or AUML sources. Due to the wide reach of AML in Appalachia, there were no readily identifiable similar sized watersheds nearby that could claim strong independence from all or most of these stresses. Monitoring at Symmes Creek was undertaken to provide evaluation of potential anthropogenic influences at the primary site on Federal Creek

3.3 METHODS AND MATERIALS

Data collection in Leading Creek and the reference watersheds included monitoring of: (1) Land use and land cover, (2) precipitation, stream stage, water velocity, and flow; (3) sediment and water quality; (4) benthic macroinvertebrate and stream habitat; and (5) toxicity through sediment, acute, and *in situ* testing programs. To assess chemical, physical, and biological interactions, stations in Figure 3.1 covered over a wide range of drainage areas and stream orders (Minshall *et al*, 1985; Strahler, 1957; Vannote *et al*, 1980; USEPA, 1996). Data collection specific to habitat, biodiversity, and land use and land cover is outlined here.

3.3.1 Land Use and Land Cover

Land use/cover data (scale 1:24000) was developed through four primary sources including a) hand-digitized 7.5-minute quadrangle series maps depicting abandoned strip mines, b) Ohio Division of Natural Resource (ODNR) geodata depicting abandoned underground mines, c) 1988 Landsat5 Thematic Mapper (TM) geodata developed by ODNR specifically for assessment of abandoned mine lands, and d) 1994 Landsat5 TM geodata developed by ODNR for NASA'a Land Cover and Land Use Change Program. The two TM data sets had resolution of 30mx30m describing watershed land use/cover characteristics. Development and qualification of all land use/cover geodata sets are presented in Appendix A.

Remotely sensed TM land use/cover variables for 1988 were described by Mine Gob (darkest coal wastes), Mine Spoil (coal waste mixed with subsoil and topsoil), Bare Soil (representing heavy agricultural activities), Sparse Vegetation (<50%), Moderate Vegetation (50 to 100%), Full Vegetation (100%, mostly herbaceous), Shrub, Wooded, Water, Wetland, Urban. Remotely sensed TM variables for 1994 were given by Urban (impervious), Ag/Urban (agricultural and open urban areas), Shrub/Scrub, Wooded, Water, Wetlands (non-forested), and Barren (mining, gravel pits, beaches, etc.). Directly developed data included AUML, ASML, and unreclaimed ASML. ASML was initially separated by disturbance before or after 1961.

Initial conversion and processing of basemap ARC/INFO® exchange file formats for ODNR geodata was carried out using either using ArcView® 3.0 and later versions or the ATLAS® Import/Export program. Data describing locations of study sites, stream order, drainage area, stream length, and associated land use/cover characteristics were calculated using ATLAS-GIS® Version 3.01 for Windows® and linked Microsoft Office® 97 spreadsheet-database programs. All raw and processed data was managed via linked ACCESS, DBASE III, and EXCEL pivot tables. All area calculations were standardized to geographic coordinates using the North American Datum (NAD) of 1927 and retained a reported positional accuracy ±15 m for TM data. Bare Soil was interpreted as agricultural activity resulting from TM images taken in June 1988, believed to be dominated by spring planting and higher density livestock operations. Full 1988 Landsat5 data coverage was not available for Symmes Creek and Federal Creek subsheds. Land use/cover data for 1994 represented less defined segregation of urban and agricultural activities and was developed from remote sensing in September and October. The 1994 ODNR coverage incorporated 1986 Ohio Wetlands Inventory geodata.

3.3.2 Habitat and Insect Community Assemblage

Biodiversity data collected in July and September 1995 during reconnaissance for this study were amended with full surveys of biota in July and September 1996, including added mainstem surveys in July, September, and October 1997. Trained, experienced technicians conducted all biological and toxicological sampling and analysis. A contract ecological services laboratory familiar with Ohio streams, fish, and insects performed sample identifications. Various biological sample collections and habitat surveys were carried out during appropriate spring and fall seasons (OEPA, 1989). For biological data collection, stations represented nearby reach areas of the stream, incorporating typical periodic features and structure defined within the banks. Riffle, pool, and shoreline habitats at each station were sampled intensely using D-framed nitex-mesh dipnets, and samples preserved in ethanol (OEPA, 1989).

Dipnet samples were sieved (0.5 mm mesh) and rinsed in the laboratory to remove ethanol, placed in an enamel pan, and separated from fine substrate for enumeration and identification. Identifications were made with the aid of various taxonomic keys (Mason 1973; Merrit *et al*, 1984; Pennak 1989; Wiggins 1977). Taxonomic values derived included total taxon richness and abundance, mayfly abundance, Ephemeroptera-Plecoptera-Trichoptera (EPT) abundance, %EPT abundance, and EPT taxon richness, among others. Qualitative EPT data were identified for all 1996 and the July and October 1997 field surveys.

Habitat sampling in June 1996 followed protocols outlined in the USEPA's Rapid Bioassessment Protocols (RBP) (USEPA, 1989). Habitat was also evaluated using Ohio EPA's Qualitative Habitat Evaluation Index (QHEI) (Rankin, 1989). USEPA's method can be seen as more of a reach-based approach where the Ohio QHEI can be viewed of more as a site-based approach. The qualitative habitat evaluations were supplemented with separate quantitative instream and riparian reach-level surveys conducted in October 1996, and June and September 1997. Measured sediment properties, channel geometry, and observations in the streambed were used to characterize geomorphologic properties from site to site. A combination of pebble counts, sieve analyses and hydrometer methods were selected for particle size analysis (ASTM, 1990, 1993; Gee *et al*, 1986; Wolman, 1954). Limited sieve and hydrometer data were provided via routine sediment toxicity testing establishing %sand, %silt, and %clay in site samples.

All statistical analysis was performed using the NCSS® 97 software package (Hintze, 1997) or routines in the Microsoft® EXCEL 97 spreadsheet package. Unless noted, a significance level $\alpha = 0.05$ was used and all Analysis of Variance (ANOVA) comparisons met tests for skewness, kurtosis, and omnibus normality, and Levine's modified test for equal variance (Hintze, 1997; Zar, 1999). Where applicable, parametric ANOVAs were supplemented using the Tukey-Kramer multiple comparison test with $\alpha_{MC} = 0.10$.

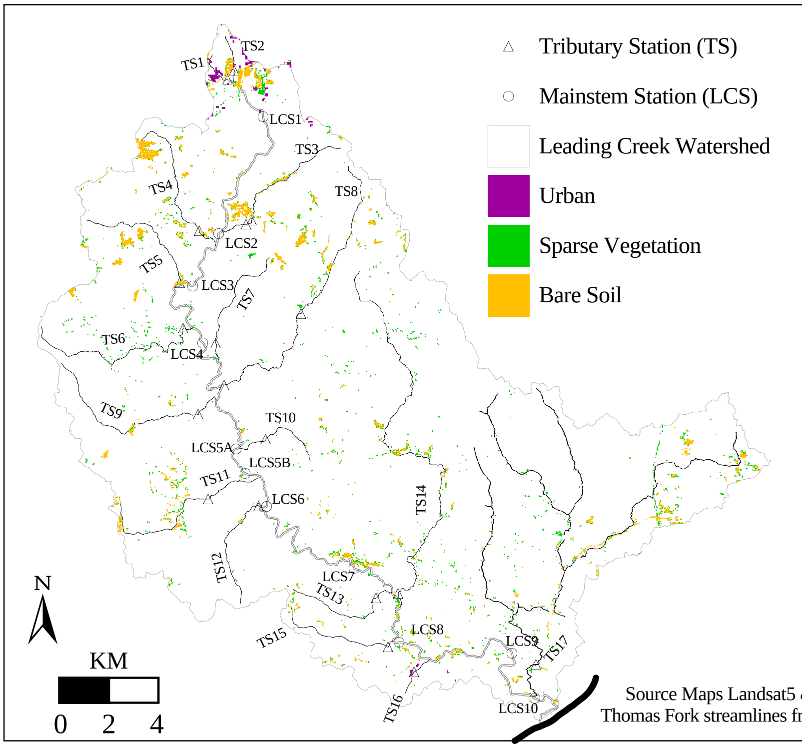
3.4 RESULTS

3.4.1 Land Use and Land Cover Analysis

Land use and land cover analysis addressing urban, agricultural and mining activities is summarized in Figures 3.2 through 3.7. Figures 3.2 and 3.3 depict urban and agricultural activity. Figures 3.4 and 3.5 depict mining activity derived from remotely sensed data. Figures 3.6 and 3.7 describe coal mining extents derived from aerial photography (ASML) or permit maps (AUML). Based on aerial photo revision dates on USGS quadrangle sheets, 80% or 1379 ha of strip mines in Leading Creek were created by 1961, and 20% or 353 ha between 1961 and 1978. Quantification of extraction of coals in Leading Creek was evaluated between hand-digitized ASML based on USGS mapping derived from aerial photography, and AUML derived from ODNR mining permit data. Excluding Thomas Fork where ASML was not electronically digitized, only 4.2% by area overlap of ASML and AUML coverage was found in subsheds with AUML>0%, indicating minimal overlap in techniques. AUML extents were centered below ridges and ASML followed outer edge contours. Adjacent extents lying outside the watershed boundary were shown in Figure 3.7. For all sites with AUML>2%, commingled ASML was also >3%, ranging 4.5% to 30%. For sites with AUML<2%, ASML ranged 0 to 11%. Figures 3.6 and 3.7 show variation of commingled AML. Bare Soil ranged 0.1% to 8.3%, Urban (1988 TM) from 0 to 13%, and Ag/Urban from 11% to 81%.

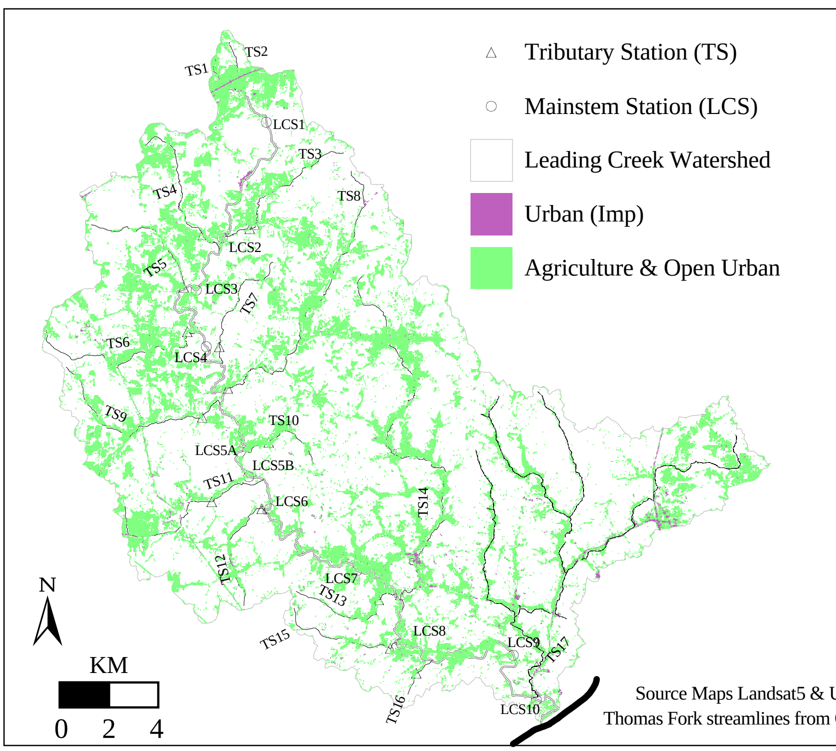
Total % of the Leading Creek Watershed for 1988 remotely sensed TM land use/cover variables were given by: Mine Gob (0.22%), Mine Spoil (0.15%), Bare Soil (1.9%), Sparse Vegetation (0.81%), Moderate Vegetation (6.5%), Full Vegetation (28.8%), Shrub (2.1%), Wooded (59.1%), Water (0.13%), Wetland (0.14%), Urban (0.12%). Excluding Mine Gob and Mine Spoil (1988 TM) associated with active mines, total % of the Leading Creek watershed would be 0.02% and 0.08%, respectively. Remotely sensed TM variables for 1994 were given by Urban (impervious – 0.68%), Ag/Urban (agricultural and open urban areas – 22.4%), Shrub/Scrub (3.3%), Wooded (72.4%), Water (0.40%), Wetlands (non-forested – 0.17%), and Barren (mining, gravel pits, beaches, etc – 0.61%). Directly developed mining LUC descriptions were defined by AUML (4.1%), total ASML (4.5%), and unreclaimed ASML (3.6%).

Urban land uses here represented a primarily rural character associated with small towns, townships, and villages. Using Meigs County data, on average, based on area approximately 7500 to 7800 residents may live within the watershed perimeter. A census of approximately 12,400 was estimated based on area and population statistics derived from Upper-Ohio Shade HUC water supply data made available from USGS. Counting population changes between 1990 and 1994 occurring within townships and villages encompassing Leading Creek, increases in population of 1049 residents were observed (on average) or equivalently a 4.7% increase was noted from the start of that decade. For comparison to Leading Creek, based on 1998 data available from USEPA for the parent Upper-Ohio Shade River Hydrologic Unit (USGS - 05030202), 1% of the parent HUC was designated as Urban and 9% by crop-use, where some overlapping can occur in these terms. Across all of Meigs County, land uses were described by average urbanization of 5%, 18% cropland, and 17% designated as pasture (USDA, 1983).



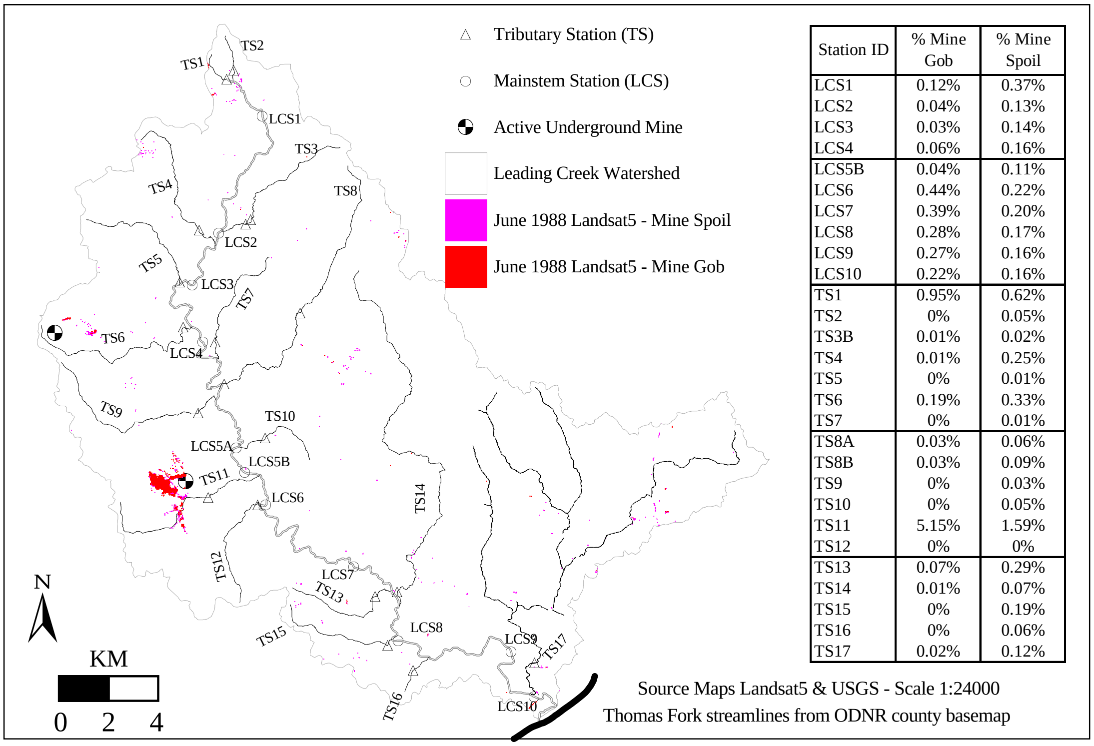
Station ID	% Urban	% Bare Soil	% Sparse Veg.
LCS1	4.0%	8.3%	2.7%
LCS2	1.2%	4.4%	1.3%
LCS3	0.9%	4.4%	1.2%
LCS4	0.5%	4.6%	1.2%
LCS5B	0.3%	2.9%	0.8%
LCS6	0.2%	2.8%	0.8%
LCS7	0.2%	2.5%	0.8%
LCS8	0.1%	2.1%	0.8%
LCS9	0.2%	2.1%	0.8%
LCS10	0.1%	1.9%	0.8%
TS1	12.7%	7.9%	2.2%
TS2	7.0%	6.7%	1.1%
TS3B	0%	3.8%	1.0%
TS4	0%	5.8%	1.2%
TS5	0%	2.9%	0.8%
TS6	0%	6.4%	1.3%
TS7	0%	0.6%	0.5%
TS8A	0%	2.3%	1.0%
TS8B	0%	1.5%	0.7%
TS9	0%	1.1%	0.3%
TS10	0%	0.1%	0.6%
TS11	0%	2.3%	1.2%
TS12	0%	0.2%	0.3%
TS13	0%	0.6%	0.7%
TS14	0%	1.0%	0.7%
TS15	0%	1.3%	0.4%
TS16	1.3%	0.2%	0.5%
TS17	0%	1.0%	0.8%

Figure 3.2 - Leading Creek Bare Soil; (June 1988 Landsat5 thematic data provided by Ohio Department of Natural Resources as part of statewide Abandoned Mine Land assessment)



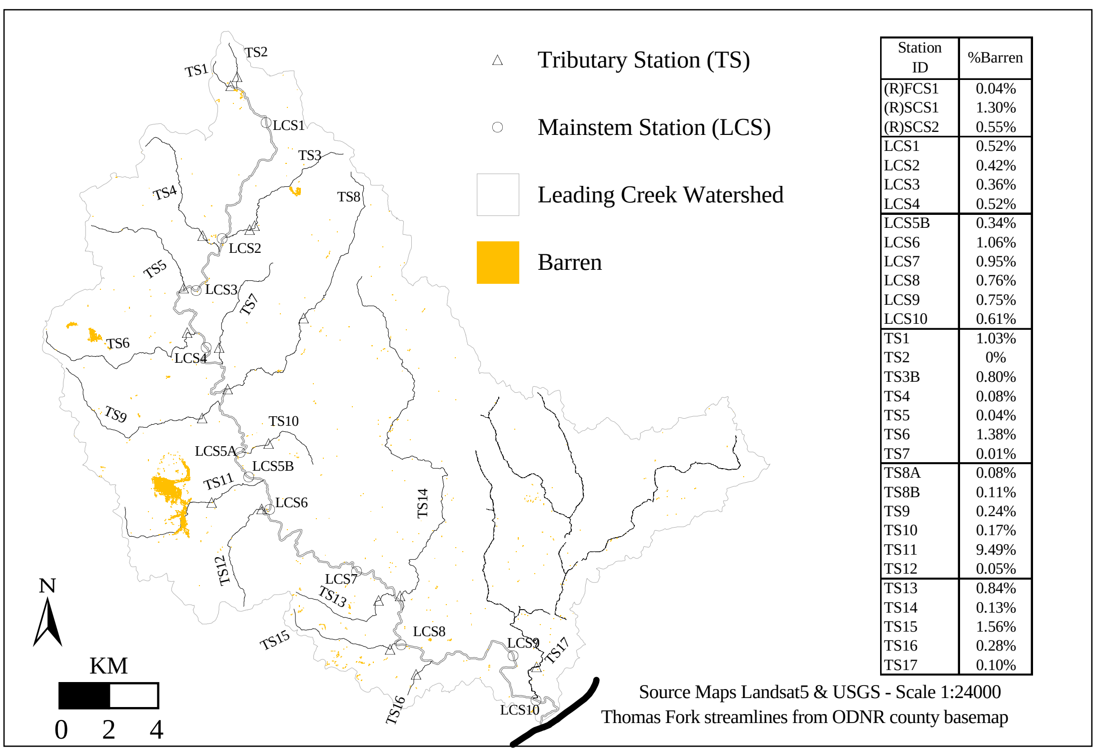
Station ID	% Urban	% AG/ Urban
(R)FCS1	0.43%	27%
(R)SCS1	1.3%	38%
(R)SCS2	0.81%	29%
LCS1	2.0%	47%
LCS2	0.95%	28%
LCS3	0.77%	30%
LCS4	0.59%	29%
LCS5B	0.44%	26%
LCS6	0.48%	25%
LCS7	0.48%	24%
LCS8	0.51%	24%
LCS9	0.50%	24%
LCS10	0.68%	22%
TS1	6.3%	81%
TS2	0.27%	40%
TS3B	0.19%	25%
TS4	0.21%	37%
TS5	0.29%	23%
TS6	0.45%	30%
TS7	0.14%	15%
TS8A	0.29%	21%
TS8B	0.25%	24%
TS9	0.34%	26%
TS10	0.11%	18%
TS11	1.1%	25%
TS12	0.42%	17%
TS13	0.39%	18%
TS14	0.56%	22%
TS15	0.60%	13%
TS16	0%	11%
TS17	1.3%	17%

Figure 3.3 - Leading Creek Agriculture/Open-Urban and Impervious Urban Areas; (Fall 1994 Landsat5 thematic data provided by Ohio Department of Natural Resources)



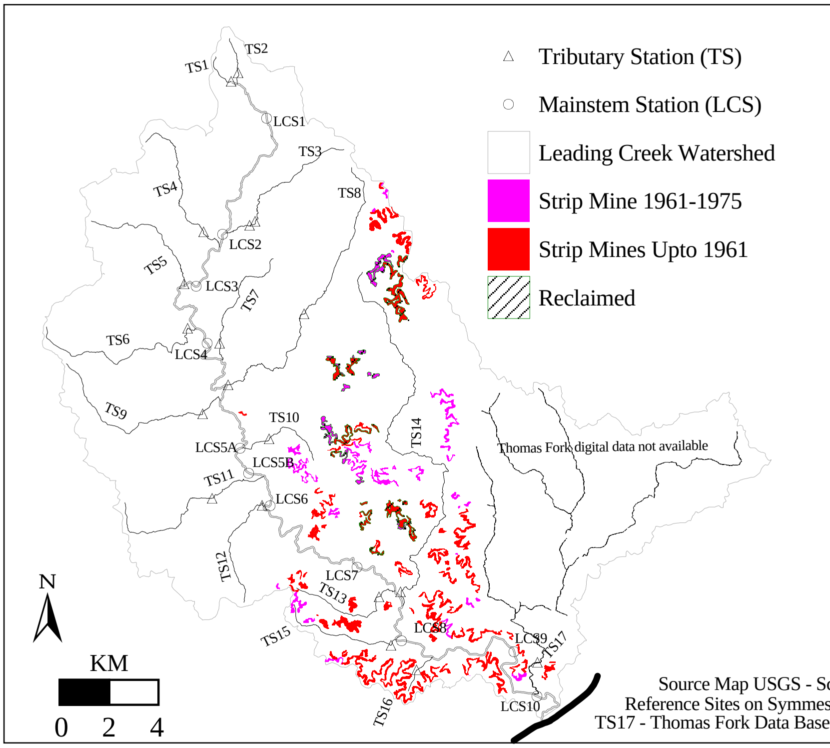
Station ID	% Mine Gob	% Mine Spoil
LCS1	0.12%	0.37%
LCS2	0.04%	0.13%
LCS3	0.03%	0.14%
LCS4	0.06%	0.16%
LCS5B	0.04%	0.11%
LCS6	0.44%	0.22%
LCS7	0.39%	0.20%
LCS8	0.28%	0.17%
LCS9	0.27%	0.16%
LCS10	0.22%	0.16%
TS1	0.95%	0.62%
TS2	0%	0.05%
TS3B	0.01%	0.02%
TS4	0.01%	0.25%
TS5	0%	0.01%
TS6	0.19%	0.33%
TS7	0%	0.01%
TS8A	0.03%	0.06%
TS8B	0.03%	0.09%
TS9	0%	0.03%
TS10	0%	0.05%
TS11	5.15%	1.59%
TS12	0%	0%
TS13	0.07%	0.29%
TS14	0.01%	0.07%
TS15	0%	0.19%
TS16	0%	0.06%
TS17	0.02%	0.12%

Figure 3.4 - Leading Creek Mine Gob and Spoil Areas; (June 1988 Landsat5 data interpretation provided by Ohio Department of Natural Resources)



Station ID	%Barren
(R)FCS1	0.04%
(R)SCS1	1.30%
(R)SCS2	0.55%
LCS1	0.52%
LCS2	0.42%
LCS3	0.36%
LCS4	0.52%
LCS5B	0.34%
LCS6	1.06%
LCS7	0.95%
LCS8	0.76%
LCS9	0.75%
LCS10	0.61%
TS1	1.03%
TS2	0%
TS3B	0.80%
TS4	0.08%
TS5	0.04%
TS6	1.38%
TS7	0.01%
TS8A	0.08%
TS8B	0.11%
TS9	0.24%
TS10	0.17%
TS11	9.49%
TS12	0.05%
TS13	0.84%
TS14	0.13%
TS15	1.56%
TS16	0.28%
TS17	0.10%

Figure 3.5 - Leading Creek Barren Land; (Fall 1994 Landsat5 thematic data provided by Ohio Department of Natural Resources - barren areas are usually associated with mining)

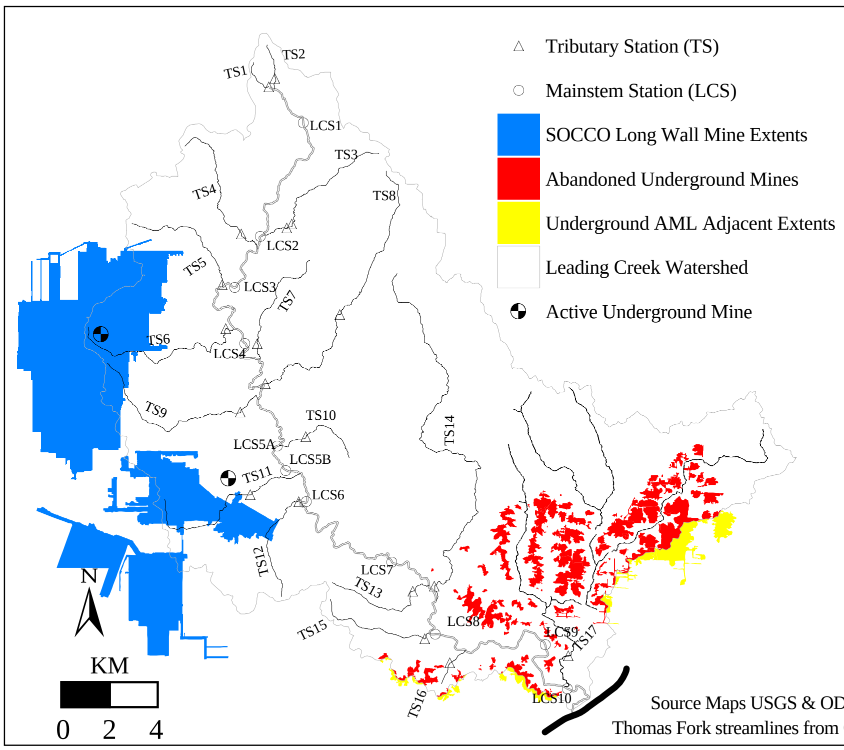


Station ID *	Total ASML	Reclaimed ASML
(R)FCS1	2.1%	0%
(R)SCS1	3.8%	0.5%
(R)SCS2	2.6%	0.3%
LCS1	0%	-
LCS2	0%	-
LCS3	0%	-
LCS4	0%	-
LCS5B	1.0%	0.4%
LCS6	0.9%	0.3%
LCS7	1.1%	0.3%
LCS8	3.3%	1.0%
LCS9	3.8%	1.0%
LCS10*	4.5%	0.8%
TS1	0%	-
TS2	0%	-
TS3B	0%	-
TS4	0%	-
TS5	0%	-
TS6	0%	-
TS7	0%	-
TS8A	5.5%	1.5%
TS8B	3.6%	1.4%
TS9	0%	-
TS10	5.3%	1.9%
TS11	0%	-
TS12	0%	-
TS13	11%	0%
TS14	8.5%	3.7%
TS15*	15%	0%
TS16*	30%	0%
TS17*	6.6%	0.3%

* - indicates AUML also >2%.

Source Map USGS - Scale 1:24000
Reference Sites on Symmes and Federal Creeks and
TS17 - Thomas Fork Data Based On Grid Counts; 1sq.=0.106ha

Figure 3.6 - Leading Creek Abandoned Surface Contour Mines; (Digitized strip mine coverages from USGS 7.5-minute quadrangle map series derived were originally from aerial photography)



Station ID *	Total AUML	Below No. 31
(R)FCS1	0.06%	-
(R)SCS1 *	0.97%	-
(R)SCS2	0.44%	-
LCS1	0%	-
LCS2	0%	-
LCS3	0%	-
LCS4	0%	-
LCS5B	0%	-
LCS6	0%	X
LCS7	0%	X
LCS8*	0.53%	X
LCS9*	0.93%	X
LCS10*	4.16%	X
TS1	0%	-
TS2	0%	-
TS3B	0%	-
TS4	0%	-
TS5	0%	-
TS6	0%	-
TS7	0%	-
TS8A*	0%	-
TS8B*	0%	-
TS9	0%	-
TS10*	0%	-
TS11	0%	X
TS12	0%	-
TS13*	0%	-
TS14*	1.67%	-
TS15*	2.79%	-
TS16*	19.5%	-
TS17*	15.9%	-

* - indicates ASML also >3%.

Source Maps USGS & ODNR - Scale 1:24000
Thomas Fork streamlines from ODNR county basemap

Figure 3.7 - Leading Creek Abandoned Underground Coal Mines; (Mine permit data digitized by ODNR as part of statewide coverage of abandoned underground mines, updated August 1999)

3.4.2 Qualitative Habitat Assessments

Raw habitat data is presented in Tables B2 and B3 of Appendix B, respectively. Tables 2.35 and 2.36 in Chapter 2 presented elemental comparisons of habitat and biodiversity scoring approaches used, respectively, in the QHEI, RBP Level II, and by Richards and Host (1994b). Level II applies to biological sampling where associated RBP habitat assessments can be generic across levels. Figure 3.8a compares Ohio QHEI versus USEPA RBP approaches and showed fair agreement for total scores site to site (R^2 = 0.62). Figure 3.8b compares total USEPA RBP scores to a reduced order 4-variable RBP score based on scour/deposition, bank vegetation, pool/riffle-run/bend, and streamside subscores.

3.4.2.1 Supporting and Non-Supporting Habitat

Compared to the reference site at Federal Creek, non-supporting or partially supporting habitat (USEPA, 1989) found for AUML>2% was comparable to scores for the group of sites with ASML>3%. Based on the reduced order 4-variable model devised here for USEPA RBP data in Figure 3.8b, site TS16 had supporting habitat, but only partially supporting QHEI habitat. In Figures 3.8a and 3.8b, non-supporting habitat for sites with ASML≤3% and ASML=0 were also identified where these sites are dominated by agricultural and urban land uses. For USEPA RBP scores, all AG-Urban dominated tributaries (defined here as stations TS1 to TS5, TS7, TS9, and TS12) had non-supporting habitat where the reduced 4-variable model had improved partially supporting habitat for TS2 and TS9 only. Based on the QHEI scores, the AG-Urban tributaries had only non-supporting or partially supporting habitat. Mixed results were seen for mainstem sites with ASML≤3%, but lower mainstem sites with ASML>3% were all non-supporting.

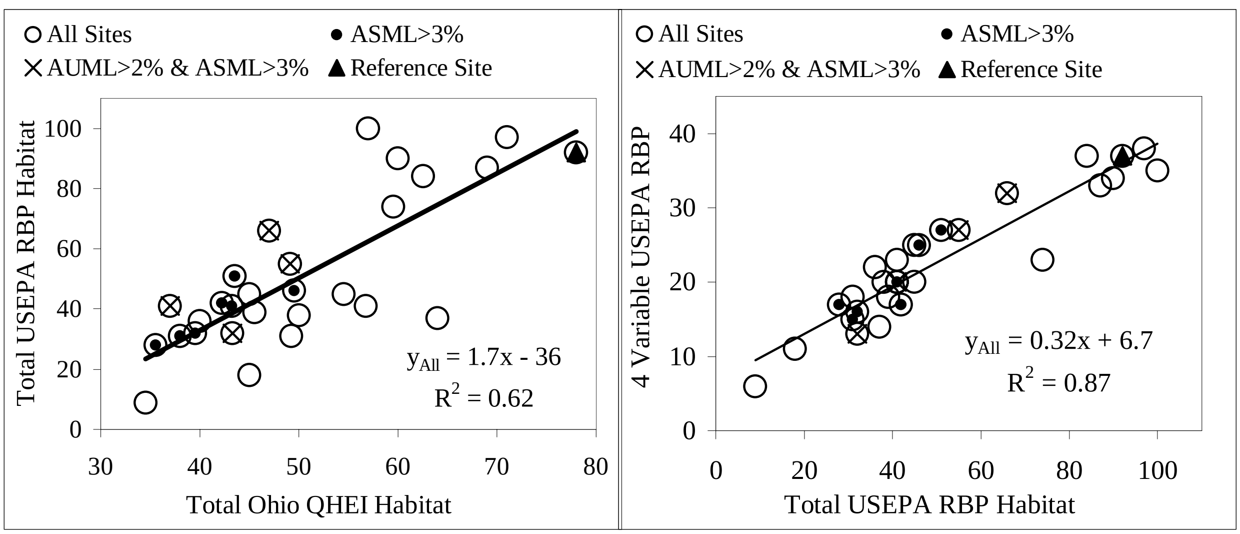


Figure 3.8a and 3.8b – 1996 Qualitative Habitat Scores (a) Total USEPA RBP versus Total Ohio QHEI and (b) Reduced 4-Variable USEPA RBP versus Total USEPA RBP

As a percentage of reference site data, non-supporting, partially supporting, and supporting habitat is defined by levels of <58%, 60% to73%, and 75% to 88%, respectively (USEPA, 1989). For the Ohio QHEI, RBP, and 4-variable RBP total scores, TS11, LCS1, LCS2, LCS6 and LCS7 were comparable to the reference site with scores ≥90% of levels at (R)FCS1.

3.4.2.2 Principal Component Analysis

Stations with AUML>2% found at TS15, TS16, TS17, and LCS10 are known to be impacted by acid mine drainage (AMD) due to low pH (Gallagher *et al*, 1999). Principal component analysis of RBP data sets with and without AUML>2% both exhibited similar constituency and weights for the first two factors identified. Examining USEPA RBP system variation without the four sites with AUML>2%, 4 of 9 factors were selected with significant eigenvalues near or greater than 1, explaining 93% of total system variation. The two strongest factors split relatively equal weights against the 3 instream and 3 physical variables comprising 34% of variation in the data. The three remaining riparian and bank structural variables dominated the second factor with cumulative variation of 57%. The third factor with cumulative variation of 82% was relatively equally weighted among six variables in strongest to weakest order given by: flow/velocity, pool/riffle-run/bend, streamside cover, scour/deposition, bank stability, and bank vegetation. Significantly less and less contribution was retained from channel alteration, substrate/cover, and embeddedness, respectively. Comparing to similar results for the data set with the four most severely AMD impacted sites retained, the two strongest factors identified retained 32% and 55% of data variation, respectively. Strongly dominated by flow/velocity, pool/riffle-run/bend, and to a lessor degree by scour/deposition, the third factor had 80% cumulative variation.

Principal component analysis of QHEI data offered little optimization capacity with 82% to 83% of variation distributed across 5 factors, with only the seventh factor falling well out of range of a unit eigenvalue for both data sets examined above. Pool and riffle sub-variables of the pool/riffle subscore were evaluated separately. The first two QHEI factors were split across instream, bank, and stream slope variables with factor one at 19% dominated by substrate, cover, channel, pool, and riffle subscores. Factor two at cumulative variation of 34% was dominated by riparian and gradient variables. The third factor reaching 51% cumulative variation was dominated by the substrate and cover variables and to a lessor degree by riffle quality. All principal components analyses used varimax factor rotation and robust covariance estimation (Hintze, 1997).

Analysis of USEPA's RBP index indicated the original nine variables shared a good deal of interdependence. Subsequent multiple linear regression was conducted using the raw habitat data set and the four strongest RBP principal component factors as dependent variables. Scour/deposition, bank vegetation, pool/riffle-run/bend, and streamside cover were the most influential variables retaining 89% to 92% of variation in the original 4 factors identified. A final principal component analysis indicated the four subscores each contributed 24% to 25% of total variation in 4 non-reducible factors where each variable had a communality of 82% to 87% associated with each of the final factors. Similar results were again obtained by exclusion of the 4 most severely impacted AMD sites with AUML>2%. Retaining all QHEI scores, further RBP analysis concentrated on the four dominant subscores identified for this protocol, and total RBP.

3.4.3 Benthic Macroinvertebrate Biodiversity

Spring, summer and fall reach results (riffle+run+pool) and site averages for qualitative dipnet collections of benthic macroinvertebrate are given in Table B1 in Appendix B. Tributary and mainstem sites were sampled twice in 1996; Parker Run and mainstem sites three times in 1997; and all sites below Mine 31 twice in 1995. Biodiversity metrics included qualitative benthic macroinvertebrate taxon richness and abundance data, for total and EPT groups, respectively. Qualitative data was used here to extend evaluations to a wider range of drainage areas and stream orders otherwise not possible with quantitative data. Quantitative collections were conducted in July and September of 1995, and July and October 1996 on lower and upper mainstem sites, respectively, using multiple-plate Hester Dendy samplers colonized for 6 weeks. Quantitative macroinvertebrate sampling on the mainstem was often compromised or invalidated by insufficient flow or sedimentation that buried samplers. Qualitative dipnet sampling performed in accordance with State of Ohio procedures (OEPA, 1989) reduces bias in estimation of taxa richness and abundance by rigorous sampling of all nearby riffle, run, and pool zones.

3.4.4 Habitat-Biodiversity Relationships

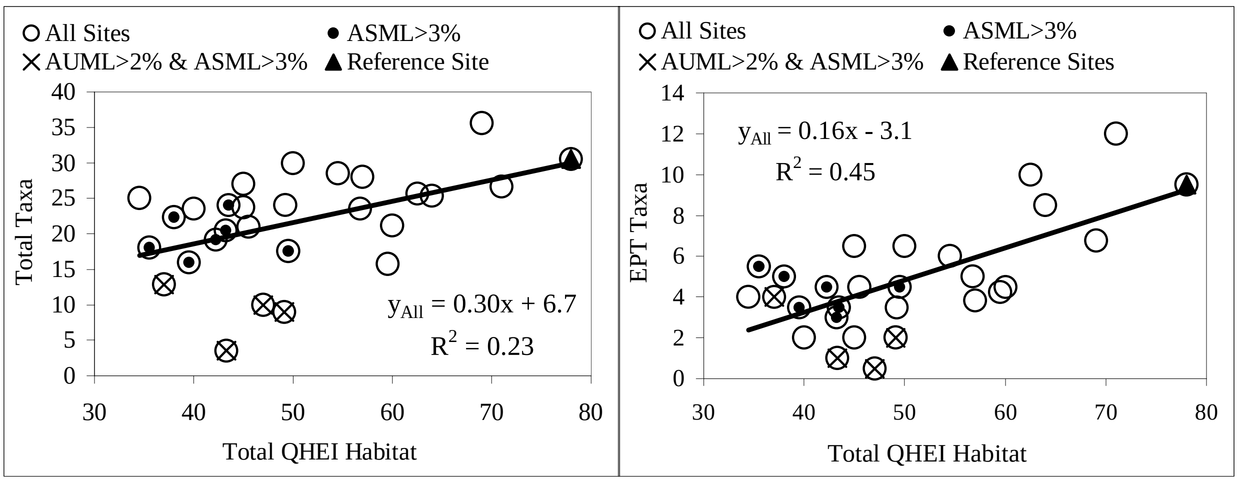
Table 3.1 summarizes spearman-rank correlations found in Leading Creek and reference watersheds between benthic macroinvertebrate biodiversity and qualitative habitat. Data was broken down by examining sites with and without severe AMD, identified by various signatures in water quality and generically as AUML>2% and AUML<2%. Habitat data with and without moderate (>2%AUML) and severe (>10%AUML) AMD impacts were examined separately to assess any differences linked to associated water quality degradation. In Table 3.1, the strongest most consistent relationships between habitat and biodiversity, with and without AMD present, were realized for riffle, substrate, Total QHEI, and channel parameters. Using a modified set of metrics, Richards and Host (1994b) also found that substrate and woody debris characteristics had the strongest correlations to macroinvertebrate assemblage and composition. Looking at the group of 4 sites with AUML>2%, no significant correlations were found except between bank vegetation and total abundance with r_s=1, and pool-riffle and both EPT scores where r_s=-1.

Table 3.1 – Spearman-Rank Correlation Coefficients for Habitat and Biodiversity;
(Blank data indicates significance level [P] >0.05; n = 24-26 AUML<2%, n = 28-30 all sites)

Biodiveristy or Habitat Parameter	Total Benthic Macroinvertebrate				EPT Benthic Macroinvertebrate			
	Abundance		Taxa		Abundance		Taxa	
	All Sites	AUML<2%	All Sites	AUML<2%	All Sites	AUML<2%	All Sites	AUML<2%
Total Taxa	0.82	0.89	-	-	-	-	-	-
EPT Abundance	0.48	0.48	0.52	-	-	-	-	-
EPT Taxa	0.52	0.52	0.64	0.50	0.83	0.76	-	-
Total RBP	-	-	-	-	-	0.44	-	-
Scour/Depostion	-	-	-	-	-	-	-	-
Pool/Riffle-Run/Bend	0.42	0.48	-	-	0.41	0.52	-	0.47
Bank Vegetation	-	-	-	-	-	-	-	-
Stream.Cover	-	-	-	-	-	-	-	-
Total QHEI	0.54	0.53	0.51	0.51	0.58	0.63	0.53	0.58
Substrate	0.41	0.43	0.46	-	0.70	0.65	0.50	0.42
Cover	-	-	-	-	-	-	-	-
Channel	0.46	0.45	-	0.45	0.45	0.57	-	-
Riparian	-	0.44	-	0.47	-	-	-	-
Pool	-	-	0.41	-	0.45	-	0.55	0.51
Riffle	0.47	0.54	0.46	0.46	0.67	0.71	0.52	0.56
Pool & Riffle	0.51	0.54	0.53	0.48	0.62	0.64	0.64	0.66
Gradient	-	-	0.38	-	-	-	-	-

Sites with AUML>2% were all insignificant except bank vegetation and pool-riffle.

Shown in Figures 3.9.a and 3.9b, QHEI habitat quality was slightly more correlated with EPT taxa than with total taxa. In Figures 3.8 and 3.9, sites with ASML>3% or AUML>2% were clearly identified as having poor overall habitat scores, though poor habitat was comparable to many sites with different land uses. Similarly, ASML>3% was not readily identified with sites with depressed biodiversity compared to sites with ASML≤3%. Thus sites with AUML>2% were clearly identified as candidates for poor biodiversity but this was not attributable to habitat. In Figures 3.9a and 3.9b AUML>2% was more easily distinguished from other sites for total taxa than for EPT Taxa. Figures 3.8, 3.9 and Table 3.1 outline a structural relationship between AUML, ASML, and aquatic habitat and biodiversity. Results underscored that AML (AUML+ASML) alone did not define all significant anthropogenic impacts in Leading Creek.



Figures 3.9a and 3.9b – Qualitative Macroinvertebrate (a) Total and (b) EPT Taxa Richness versus Total Qualitative QHEI Habitat Scores

Only modest differences between habitat and biodiversity were observed in Table 3.1 comparing data sets with and without severe AMD from AUML>2%. Stations TS15, TS16, TS17, and LCS10 also had poor or average results for the strongly correlated habitat parameters identified in Table 3.1 with ASML<2% and for all sites. Levels of poor habitat noted for AUML>2% and ASML>3% were typically comparable to the group of sites with ASML>3% and AUML<2%. Habitat-biodiversity correlation analysis could not discern between lack of biodiversity attributed to poor habitat or poor water quality at AUML>2%, but did indicate that poor habitat is responsible in part for depressed EPT biodiversity at sites without severe AMD. Spearman-rank analysis indicated that reduced pool-riffle subscores within AUML>2% were not attributable to increasing levels of AUML, but showed that poor water quality elicited strong positive correlation between biodiversity and bank vegetation. Figure 3.9a exposes the weak influence habitat has on overall biodiversity across Leading Creek, where EPT data exhibited mixed results except for sites with exceptional habitat.

While ASML appeared to impart effects upon habitat at sites with AMD and AUML, as at sites with high rates of %ASML but little AMD and AUML, the added presence of severe AMD did not negatively influence habitat scores above levels attributable to ASML. One possible difference that might be expected in severely AMD impacted reaches is a higher degree of siltation due to oxy-hydroxide precipitates. This could effect 25% of the total substrate score that itself represents 20% of the total QHEI. Riffle quality represents 8% of the total QHEI score and comparing AUML to ASML appeared to be improved. No distinction could be made here between surface disturbances tied to AUML, and not ASML, for example in the form of mine spoil, mine gob, or mine dumps. True for Leading Creek, an assumption that the aereal extent of mining land surface disturbance is not strongly correlated to the severity of AMD was needed to conclude differences in impact upon habitat from AUML-AMD and AUML/ASML activities.

Substrate and riffle degradation shown in Table 3.1 were strongly indicative of accelerated anthropogenic induced erosion and instream sedimentation which are known to be common problems in most streams impacted by either mining or agriculture (OEPA, 1991a, 1991b; USDA, 1983, 1985, 1998). QHEI riffle-run quality addresses depth, stability, and extent of riffle embeddedness. Both QHEI substrate type and quality can also be radically altered by fine material inundation (Diplas, 1994; Richards *et al*, 1994a; Shirazi, 1981) since the indices incorporate the degree of pavement siltation and subpavement embeddedness. The 4-variable RBP data set evaluated in Table 3.1 was relatively weak for habitat-biodiversity relations except for total and EPT abundance pool/riffle-run/bend scores. The RBP pool/riffle-run/bend index scores the quality of the strongest feature subgroup (e.g. pool/riffle) and is similar to attempts in the QHEI to rate geomorphologic development. The premise is that complexity and diversity in fluvial morphology leads to similar attributes in biotic communities (USEPA, 1996).

3.4.5 Landscape-Habitat Relationships

Table 3.2 provides spearman-rank correlation coefficients between total qualitative habitat scores and land use/cover data. Using habitat variables significantly correlated to biodiversity, correlation analysis of land use/cover variables compared to habitat variables gave mixed results. Similar to habitat-biodiversity relations, landscape-habitat relations were also weak. There were three main features of interest. The first was that deleterious land uses (-) were all associated with mining or 1988 and 1994 woodlands. The woodland categories often were co-located with historic mining where each was modestly correlated to ASML with correlations of 0.62 to 0.70. Comparisons shown in Figures 3.8 and 3.9 also indicated consistently low habitat scores for ASML. Because a mixture of small to large tributaries and medium to large mainstem stations were considered, correlations did not appear to be spuriously related to morphological structure. Residents of Leading Creek consistently reiterated the loss of pools, some >3 m deep in the mainstem and in some tributaries. Deeper pools were apparently more prevalent before 1950 and the onset of strip mining and more widespread agricultural activities that exist today.

Based on interviews with older residents and multi-decade sediment depth surveys conducted by USDA in Leading Creek, sand size particles dominate sediment beds (1-2m thick) in the strip mine district compared to undisturbed conditions. These perceptions of change might be dampened by late 19th century underground mining influences on habitat that degraded the background reference point. AUML may have lead to significant sedimentation prior to later rises in ASML. Previous work in Leading Creek found that relative depth of bed sediments, by characterization of a crude modified-standard penetration test, was negatively correlated to total taxa (R^2 =0.43) (Gallagher *et al*, 1999). Massive sediment depth build-up represents a known causal link to impaired habitat-biodiversity relations due to shifting sands that adversely impact benthic macroinvertebrate population stability (Allan, 1995; Richards *et al*, 1994a, 1994b).

Table 3.2 – Spearman-Rank Correlation Coefficients for Land Use/Cover and Habitat; (Blank data indicates significance level [P] >0.05; n = 27-28; URASML = unreclaimed ASML)

Land Use/Cover Category	USEPA RBP Level II		Ohio EPA Qualitative Habitat Index					
	Total RBP	Pool/Riffle-Run/Bend	Total QHEI	Substrate	Channel	Pool	Riffle	Pool & Riffle
Directly Digitized LUC								
%ASML	-	-	-	-	-	-0.46	-	-0.46
%UnReclaimed ASML	-	-	-	-	-	-0.45	-	-0.45
%AUML	-	-	-	-0.43	-	-	-	-
%AUML+%URASML	-	-	-	-	-	-0.46		-0.45
Below Mine No. 31	-	0.39	-	-	-	-	-	-
1988 Landsat5								
%Mine Gob	-	-	-	-	-	-	0.52	0.40
%Mine Spoil	0.40	0.47	-	-	-	-	0.47	0.41
%Gob (No SOCCO)	-	-	-	-	-	-	-	-
%Spoil (No SOCCO)	-	-	-	-	-	-	-	-
%Bare Soil	-	-	-	-	-	-	-	0.42
%Sparse Vegetation	-	-	-	-	-	-	-	0.39
%Mod. Vegetation	-	-	-	-	-	-	-	0.40
%Full Vegetation	-	-	-	-	-	0.38	-	-
%Shrub	-	-	-	-	-	-	-	-
%Wooded	-	-	-0.39			-0.42	-	-0.48
%Water	-	-	-	-	-	-	-	-
%Wetland	-	-	-	-	-	-	-	-
%Urban	-	-	-	-	-	-	-	-
1994 Landsat5								
%Urban	-	-	-	-	-	-	0.38	-
%Ag/Urban	-	-	0.42	-	-	0.47	-	0.51
%Shrub	-	-	-	-	-		-	-
%Wooded	-	-	-0.41	-	-	-0.45	-	-0.50
%Water	-	-	-	-	-	-	-	-
%Wetland	-	-	-	-	-	-	-	-
%Barren	-	-	-	-	-	-	-	-

The second feature observed in Table 3.2 was that typical indicators of agricultural/urban activities were all positively correlated to pool quality. The Ag/Urban cover category was also determined to be significant in the case of total QHEI. The cause and effect in these correlations might be questioned in context of mixed results for Ag/Urban land uses shown in Figures 3.8 and 3.9. Bare Soil that was significantly (α = 0.05) correlated to Ag/Urban (r_s = 0.90), when both of these categories were negatively correlated to %ASML (r_s = -0.62 to -0.60, respectively) and %AUML (r_s = -0.45 to –0.36). These correlations appeared to express a mirror-image relation for woodlands. The results can be explained by significant mining that occurred on steeper hillsides and ridge tops where woodlands dominate the associated landscape, and where Ag/Urban activities are less prevalent. This was graphically distinguished in Figures 3.2, 3.3, 3.4 and 3.7, and in Figures A1 and A2 in Appendix A, and in county soil surveys.

Thirdly, the proximity and relative strength of Mine Gob and Mine Spoil from Parker Run in Figure 3.4 was associated with three sites TS11, LCS6 and LCS7, each with comparable habitat to the reference site. The association of beneficial impact from Gob and Spoil dominated by SOCCO's managed waste pile is most likely not causal. This was supported by comparison tests for Gob and Spoil without SOCCO operations that were insignificant for all habitat scores. The 1994 Landsat5 Barren category was correlated to both Mine Gob and Mine Spoil ($r_s = 0.71$ to 0.74, respectively), but not if SOCCO data was removed. Finally, as Table 3.2 illustrates, no other significant correlations were observed for other land use/cover metrics. Discussed for land use/cover categories in Tables 3.1 and 3.2, it is emphasized that all comparisons of agriculture incorporate some urban aspects. This was true for all available qualitative and quantitative metrics, but TS1, TS2, LCS1 and (R)SCS1 were distinguished by increased urban activity based on %Urban data. Important to interpretations here, all subsheds mentioned that are impacted by AUML have ASML>3%. Reference sites have minor percentages both of AUML and ASML.

3.4.5.1 Landscape-Habitat Multiple Test Comparisons

Directly comparing average total QHEI for the 2 tributary sites impacted by active underground mines, 11 mainstem and tributary sites impacted by agriculture, and 14 sites with AML, a significant difference was observed only between the reference site Federal Creek and AML sites ($n = 28$, [$P = 0.035$]; $\alpha = 0.05$, $1\text{-}\beta = 0.69$). No significant difference in total QHEI scores could be seen though in comparison of 8 tributary sites impacted by AML to 8 tributary sites without AML ($n = 16$, [$P = 0.20$]; $\alpha = 0.05$, $1\text{-}\beta = 0.24$). No differences were also noted in total QHEI between 8 tributaries without AML, and 14 mainstem and tributary sites with AML ($n = 22$, [$P = 0.74$]; $\alpha = 0.05$, $1\text{-}\beta = 0.06$). For the latter comparisons, Ogden Run (TS6) was included with agricultural tributaries, but TS11 and Federal Creek were excluded. Weak correlations and lack of significance among land uses above indicated that a threshold level of land use was needed to discern significant impact.

The threshold land use concept was directly tested via one-way ANOVA utilizing a distinction that balanced available data for Ag/Urban and Bare Soil against ASML mining activity. This was done realizing that all subsheds studied with ASML<3% also had Bare Soil<3%, while all tributaries with ASML had levels > 3%. The 3% by area level appeared to be a reasonable threshold to test the significance of agricultural pollutant loading given the number of sites without ASML influence. An ANOVA design ($\alpha = 0.05$) employing Tukey-Kramer multiple comparisons was constructed testing the null hypothesis of differences between means among subsheds split by thresholds of Bare Soil=3%, ASML=3%, and AUML=2%.

In the final experimental design, lower mainstem sites from LCS5A to LCS7 were classified as Bare Soil<3%, where these sites also were distinguished by ASML< 3%. To evaluate the possible association of urban influences on small headwater areas, a threshold of Urban=5% (1988 TM) was also examined, along with initial segregation lumping active underground mine influenced tributaries TS6 and TS11, and separating the reference site Federal Creek. Results of comparisons against responses in habitat scores listed in Table 3.2 are summarized in Table 3.3 and Figure 3.10. The table also lists the range of drainage areas associated with each group. Separate testing for sites having mixed land uses of ASML<3% and Bare Soil<3% were avoided due to lack of tributary representation, complete serial interconnection on the mainstem, and to minimize comparisons. For statistical comparisons in Table 3.3, power 1-β ranged from 0.89 to 1.00 for significant tests, 0.67 to 0.71 for insignificant QHEI tests, and 0.34 (total RBP) to 0.51 for insignificant RBP comparisons.

Table 3.3 –Parametric ANOVA Results for Land Use/Cover-Habitat Relationships; (For significant tests, significant differences between groups are identified by group #'s in {})

Group #	Land Use/Cover Category	# Sites	Average RBP Habitat		Average Ohio EPA Qualitative Habitat Index						Range of Drainage Areas (km^2)
			Total RBP	Pool/Riffle-Run/Bend	Total QHEI	Substrate	Channel	Pool	Riffle	Pool & Riffle	
1	Reference Site *(R)FCS1*	1	92	11	78.0 {2,3,4}	14 {2}	20	9	6.5	15.5 {3,4}	67.9
2	Urban >5%	2	40.5	2.0	42.5 {1}	1.0 {All}	11.5	5.0	0.0	5.0 -	1.2 - 1.7
3	ASML > 3% *(w/ AUML < 2%)*	7	38.7	4.4	41.6 {1,5}	7.7 {2,6}	11.9	2.6	0.7	3.2 {1,5}	4.6 - 303
4	AUML > 2% *(w/ ASML >3%)*	4	48.5	5.8	44.1 {1}	7.5 {2}	13.0	2.8	0.5	3.3 {1}	1.6 - 387
5	Bare Soil > 3% *(w/ Urban<5%)*	6	50.8	5.2	57.0 {3}	10.0 {2}	13.8	6.5	2.6	9.2 {3}	9.7 - 86.1
6	BareSoil < 3% *(w/ ASML<3%)*	6	56.7	7.0	54.2 -	11.3 {2,3}	14.7	6.5	1.8	8.3 -	6.1 - 209
7	Active Mines	2	72.5	11	55.8 -	11.3 {2}	14.5	7.0	3.5	10.5 -	18.9 - 14.8
All	([P]; α = 0.05)	28	0.39	0.17	0.036*	0.00013*	0.072	0.055	0.061	0.011*	1.2 - 387

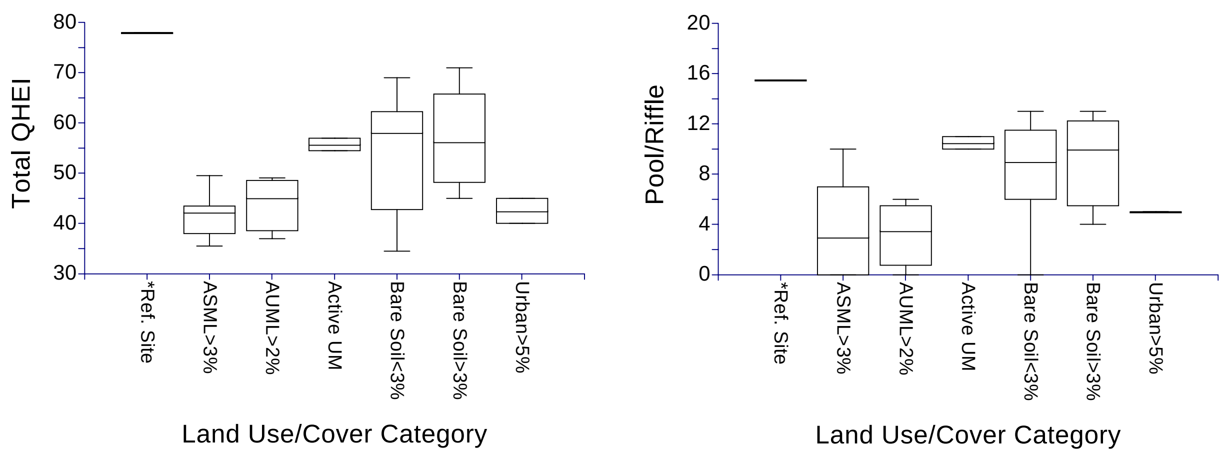


Figure 3.10a and 3.10b – Land Use/Cover Comparisons for (a) Total and (b) Pool/Riffle Qualitative QHEI Habitat Scores

Shown in Table 3.3 and Figure 3.10a Total QHEI scores were significantly lower if Urban>5%, ASML>3%, or AUML>2% compared to reference site data, with average scores of 42.5, 41.6, 44.1, and 78, respectively. At 57.0, total QHEI values at sites with Bare Soil>3% were also found to be significantly higher than sites with ASML>3%, but were not significantly lower with respect to the reference site. Average total QHEI scores at sites with Bare Soil<3% and sites with active mining, 54.2 and 55.8, respectively, were accepted as being similar in magnitude to the reference site due to increased variation and small sample sizes associated with those groups, respectively. Except for the urban land uses, similar differences between groups was observed for averages results found for the QHEI pool/riffle subscore. ASML>3% and AUML>2% had the lowest average values, 3.2 and 3.3 respectively, compared to the reference site value of 15.5.

In contrast, the Urban>5% QHEI substrate subscore was significantly different from all other land use groups with an average value of 1.0 compared to the reference site value of 14. A minor, but nonetheless significant difference was also noted for substrate between ASML>3% and Bare Soil<3% with values of 7.7 and 11.3 respectively. For remaining strong habitat-biodiversity relations in Table 3.1, no significant differences between any groups in Table 3.3 were recorded for QHEI subscores channel, or separate renditions of pool or riffle associated with the pool/riffle score. Equally, no differences were found among average total RBP or pool/riffle-run/bend scores.

Compared to reference data, significantly lower values among ASML>3% and AUML>2% land uses were observed for total and pool/riffle QHEI scores. QHEI substrate scores for Urban>5% were also significantly lower than all other land uses/cover categories.

3.4.5.2 Particle Size Characteristics

Pebble counts (Wolman, 1954) were completed on 21 cross-sections from headwaters areas downstream to LCS7. Below LCS7 gravel beds begin to transition to sand bed reaches. Representing biased (Fripp *et al*, 1993) estimators, average D_{50} and D_{90} were 24.8mm (σ = 14.5) and 58.6mm (σ = 34.6), respectively. Three of four sites had D_{90} ≥110mm believed to be related to human activities including bridge building and creating livestock crossings. Figure 3.11 shows variation in particle size fractions for fine material <2mm in gravel-bed and sands beds at routinely monitored tributary and mainstem sites. Fine material determined by hydrometer testing was classified as sand >0.075 mm, and silt/clay < 0.075 mm. In assessing fine particle size fractions (Yang, 1996), land use was contrasted between urbanization, ASML and agricultural activities.

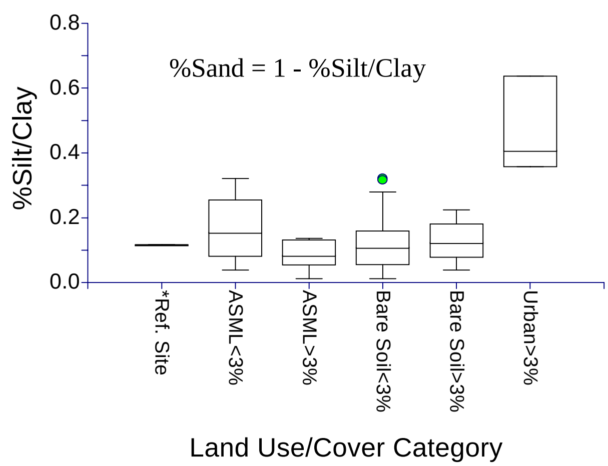


Figure 3.11 - Land Use/Cover Comparisons for Particle Size Fractions of Material <2 mm: Contrasting Urbanization Against Abandoned Strip Mines and Agriculture

Parametric one-way ANOVA testing with Tukey-Kramer multiple comparison testing was again employed for fine particle fraction analysis with α_{MC} = 0.10, where tests met assumptions for normality and equal variance (Hintze, 1997; Zar, 1999). Results showed that significant differences in silt/clay content were observed in small heavily urbanized subsheds. This was first tested by evaluating strip mining land uses above and below 3% land surface area disturbance (n = 28, [P = $2x10^{-6}$]; α = 0.05, 1-β = 1.00). Significant differences in agricultural land uses were also tested, defined by remotely sensed Landsat5 Bare Soil = 3% (n = 27, [P = $3.1x10^{-5}$]; α = 0.05, 1-β = 1.00). For this latter test, skewness normality of residuals was borderline rejected with [P] = 0.0497. In multiple comparison testing, Urban>3% was significantly different from undistinguished differences in agricultural activities. LCS1 may be impacted by siltation from closed quarry operations, though this may be insignificant since land surfaces have been reclaimed for greater than 10 years. Station TS11 below active Mine No. 31 and two small agricultural subsheds, TS9 and TS12, all with Bare Soil<3% exhibited low %silt/clay with values of 32%, 28%, and 32%, respectively. Spearman-rank and Pearson correlation coefficients between %silt/clay and substrate habitat scores though were insignificant (α = 0.05; [P]=0.60, and [P]=0.078, respectively), where TS1 and TS2 showed unusually low scores compared to all sites including TS11, TS9 and TS12.

For urban contrasts with strip mining activity, multiple comparison testing with the reference site indicated similar results. Placing the single external reference watershed station (R)FCS1 (ASML=2.1%) within strip mining groups though indicated a significant difference also existed with lower %sands at sites with ASML<3%, and higher %sands with ASML>3%. Fine sediment samples from gravel beds of sediments were not volumetrically equivalent with respect to those beds, qualifying these comparisons. Gravel beds were not observable for the most part in cross-sections sampled for any subshed with ASML>3%. Data for ASML>3% not only distinguished relative fine material content with increased sand fractions, but also absolute substrate characteristics, indicating the severity of sand inundation for ASML>3%. Increased sand fractions for AML>3% are derived from sandstones exploded from ridge tops during coal extraction. While smaller subsheds retain gravel-bed hydraulic control at points downstream, all bed forms present would best be characterized as partially to completely inundated by fine sands (Diplas *et al*, 1992).

A distinction in particle size analysis here was that station LCS1 (Urban = 4%) directly below the quarry and two small urbanized headwaters subsheds were classified together as urbanized by lowering the previously used threshold from 5% to 3%. Looking at Urban>5%, similar results were obtained with more gradual delineation of differences between groups due to increases in silt/clay content for ASML<3% and Bare Soil>3%. Looking at strip mining, a clearer delineation of ASML>3% having the greater sand content than ASML<3% was established.

3.5 DISCUSSION

This work investigated landscape looking at how habitat relates to metrics of aquatic biodiversity and watershed land use/cover variables. A 388 km^2 watershed heavily impacted by various mining techniques was assessed. Ideal for study of AML mechanisms in a watershed setting, anthropogenic impact in Leading Creek is segregated, becoming increasingly complex downstream with the addition of different non-point sources. Shown in Figures 3.1, 3.2 and 3.3 pollutant sources include mixed agricultural and urban influences, the heaviest of which occur upstream. Active underground mine point source treated effluents enter midstream. Described in Figures 3.4 to 3.7, mild historical AML sedimentation and untreated AMD impacts also join midstream, becoming severest farther downstream.

A watershed scale ecological risk assessment is significantly enhanced by an informed instream/riparian survey design that can be quantitatively related to watershed attribute data for land use/cover. Several convergent methodologies for conducting large-scale instream and riparian surveys have developed over the last two decades (Delong *et al*, 1991; Karr, 1991; OEPA, 1989; Rankin *et al*, 1989; Richards *et al*, 1994b; USEPA, 1989). Most have in some way attempted to facilitate a more informed and sometimes quantitative based ecological assessment of habitat structure. Habitat assessment methodologies presented by Ohio EPA's QHEI score and USEPA's standard RBP studied here attempt to integrate multi-disciplinary assessment measurements and associated endpoints commonly used in watershed ecological risk assessment. One difference between USEPA's RBP and the Ohio QHEI score is that the QHEI is more of a site based score, where the RBP analysis is more of a reach based assessment. The RBP incorporates typically a longer stream length for assessment of upstream macro-scale physical and structural parameters.

By comparing individual and combined impacts from each mining operation against other areas not mined, mining, agricultural and urban impacts were analyzed for differences in aquatic habitat and biodiversity. To place habitat and land use/cover into context, similar to the work of Richards and Host (1994b), the first task was to understand which habitat variables were strongly correlated to biodiversity. Significant habitat-biodiversity relations were then evaluated in context of landscape stressors, looking at relationships between habitat and land use/cover at scales of 1:24000. Richards and Host's (1994b) approach examined functional metrics of macroinvertebrate biodiversity in addition to total taxon and EPT taxon richness examined here. For example they looked at filters and tied agriculture to embeddedness. The small select set of broader based biodiversity metrics chosen here was more easily managed with an expanded list of land use/cover variables. Landscape variables were assessed as percentage of catchment area.

3.5.1 Landscape-Habitat Relationships

At a scale of 1:24000, significant results were obtained assessing ecological risk of poor habitat among subwatersheds due to AML and urban activity when poor habitat was also correlated to poor biodiversity. The work also showed that significant correlation of poor habitat to poor biodiversity for all RBP and some QHEI subscores considered were not always consistently attributable to land use/cover categories and thresholds devised here.

Evaluation of USEPA RBP qualitative habitat assessments indicated that 4 of the 9 variables dominated system variation, including scour/deposition, bank vegetation, pool/riffle-run/bend, and streamside cover. Ohio QHEI data indicated all variables were important in retaining watershed variation expressed in raw data. Conclusions from analysis of qualitative habitat data indicated that little difference in reduced order models of qualitative habitat could be observed between sites with and without severe AMD impact. No single habitat score showed large differences between mining and AMD impact. Evaluation of active mining land uses indicated that total scores from QHEI and RBP qualitative habitat assessments were good compared to sites influenced by agriculture or minor strip mining activity <3% by area. Active mining, ASML<3%, and all agricultural activities compared favorably to reference sites subjected to minor AUML, ASML and agriculture activities. The highest levels of agricultural activity were insufficient to show impacts to habitat. No significant differences were discerned between %Bare Soil and habitat data that was also negatively correlated to biodiversity.

This work had success in direct non-parametric correlation analysis of habitat-biodiversity relationships. Comparing landscapes directly to habitat, significant differences among sites studied were realized between total and pool/riffle QHEI habitat scores for abandoned surface (ASML) and underground mining (AUML) land uses greater than 3%, and 2% in aereal extent, respectively. Based on a small population of sites below the Town of Albany designated by Urban extent >5% by area, substrate data was also realized to be significantly different from all other land uses examined. Sedimentation from urban and mining activities is known to be a significant impact from nonpoint sources (USDA, 1983, 1985, 1998; OEPA, 1991a, 199b). Second only to organic enrichment and dissolved oxygen depletion, it was recognized as the second leading cause of impairment to streams and lakes from all sources identified by Ohio EPA in their 1994 Ohio Water Resource Inventory (Yoder *et al*, 1996). For urban land uses contrasted with strip mining and agricultural activities, multiple comparison testing indicated urban areas had significantly higher rates of siltation, whereas areas with ASML>3% had massive inundating levels of sand over gravel beds, and relatively higher proportions of sand materials >0.075 mm in total fractions < 2mm.

While variation in scale was not studied in this work, an overall finer resolution for geodata collection was utilized to improve viability of utilizing a hydrologic-hydraulic based approach. For ecological risk assessment courser scales of investigation (e.g. 1:100000 to 1: 250000) would be expected to be less successful in associating land use/cover variables with metrics of aquatic biodiversity (Richards *et al*, 1994b). In their work, Richards and Host's found that specific habitat metrics influenced specific functional insect groups like shredders and filters and embeddedness, and in turn poor habitat could be correlated to specific land uses (e.g. agriculture and urban activities) at course map scales of 1:250,000. This work, done at a finer landscape scale (1:24000) and using broader biological indices of total and EPT macroinvertebrate, found that total QHEI habitat metrics including subscores for pool/riffle, substrate, and channel parameters were most negatively correlated to reduced biodiversity.

Poor habitat expressed in lower total QHEI scores and subscores for substrate and pool/riffle were also significantly associated with urban and strip mining activities, but not agriculture. Unlike Richards and Hosts findings, Leading Creek landscape-habitat and landscape-biodiversity relationships were not significantly influenced by agricultural activities compared to reference site data with relatively good habitat. This was true also if mining land uses with ASML>3% were excluded from test populations. Levels of Bare Soil <3% and >3% were evaluated for differences but none were found. This was unexpected for areas visually influenced by agricultural encroachment of riparian zones that also had quantifiably the heaviest siltation component in the watershed, next to small urban subsheds.

Richards and Host examined only the significance (α = 0.05-0.10) of Pearson correlations in establishing habitat-biodiversity and landscape-biodiversity relationships. For example substrate embeddedness increased with increasing agricultural activity (R_P =0.63; α = 0.05), and urban development led to decreased substrate size (R_P =0.55; α = 0.10). Similarly weak relations in Spearman-rank coefficients were observed in Leading Creek, along with a structural coincidence for agricultural lands negatively correlated with woodlands, where woodlands were correlated to mining activities by proximity of coal. Results here do not contradict the work of Richards and Host. Results may simply point out that in using broader perspectives integrating multiple functional compositions in biodiversity (e.g. total taxa), agriculture does not significantly appear to impact biodiversity nor does it negatively impact specific habitat parameters studied. Moderate significant impairment compared to ASML and AUML might be realized in similar analysis if the number of reference sites were increased.

3.5.2 Directly Developed and Remotely Sensed Geodata

Use of equivalent or finer scale data from aerial photos and soil surveys may be available to better relate changes in land use/cover to changes in water quality (Fauss, 1992). Found in this work, directly derived geodata was superior to remotely sensed data in attempting to assess ecological risks to habitat attributed to ASML mining activities. This was true for Barren Lands categorization for environmental assessments developed through NASA'a Land Cover and Land Use Change Program using 1994 Landsat5 imaging. It was also the case for Landsat5 1988 geodata sets developed specifically to assess abandoned mine lands. Improved discrimination in spectral processing may be possible. Primarily through directly developed geodata based on aerial photography and metadata, this work successfully extended applications of 1:24000 scale geodata and GIS processing to ecological risk assessment frameworks for aquatic habitat.

3.6 CONCLUSIONS

For mining land uses, remotely sensed data was found to be far less reliable in predicting stress to aquatic habitat than was directly developed geodata based on aerial photography and metadata. Instream sedimentation from ASML and urbanization were severe in Leading Creek and represented significant degradation to aquatic communities. Comparison was made of two rapid bioassessment protocols for habitat assessment including USEPA's RBP and Ohio EPA's QHEI index. Results indicated the QHEI to be a more reliable indictor of increasing mining land use activity, and also found the 9-variable RBP to be reducible to 4 key variables describing 90% of total system variation. Data analysis found that strip mines > 3% by area represented moderate concern in depressed biodiversity. Like ASML, urban influences were also noted to impart significant overall degradation to habitat in smaller headwater subsheds related to poor substrate where ASML was noted for overall degradation and poor riffle/pool habitat. Since ensuing in 1978, benefits associated with reclamation of ASML and better instream habitat were difficult to perceive in this data set. Far-field sites downstream may not yet have realized benefits of erosion control upstream. Massive sedimentation characterized by deep, shifting sands exists in many areas with ASML>3%. Urban impacts were in part attributed to siltation

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Chapter 4 - A GIS BASED WATERSHED RISK ASSESSMENT OF URBAN, AGRICULTURE AND COAL MINING IMPACTS UPON AQUATIC BIODIVERSITY

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(ABSTRACT)

Coal mining in the Leading Creek Watershed is commingled with urban, agriculture, and limestone quarry operations. The impacts upon biodiversity from urban and agricultural activities were investigated along with effects from abandoned strip mine land (ASML), active deep underground mines, and abandoned near-surface underground mine land (AUML). Two active mines in Leading Creek currently produce 5.5 to 6 million tons of coal annually. GIS-based use/cover analysis was accomplished using both directly developed and remotely sensed geodata at a scale of 1:24000. This work extended landscape-habitat analysis by examining landscape-biodiversity relationships directly. Land use/cover data was based on remotely sensed Landsat5 data (1988 and 1994) and directly developed data from 7.5-minute quadrangle maps and historical AUML permit data. Threshold criteria for land use/cover were selected for study based on aereal extent. These included AUML=2% and 10%, ASML=3%, Urban=3% to 5%, Bare Soil=3%, and Boolean functions for active mining. Bare Soil was attributed to spring planting mixed with higher density livestock operations. Biodiversity was based on qualitative benthic macroinvertebrate taxon richness and abundance, for total and EPT groups, respectively.

AUML>2%, ASML>3%, and Urban>5% showed significant stresses to biodiversity where active mining operations and agriculture activities represented relatively insignificant stressors. Significant differences were observed in biodiversity for AUML>2%, and to a lessor degree for ASML>3%. Results showed total taxa and to a lessor degree EPT taxa were the most sensitive biometrics for both AML land use/cover categories. Stress to biodiversity from AUML>2% was attributable to AMD and conditions with low pH. Impact from ASML>3% was attributable to degraded instream habitat and not AMD. EPT taxa and abundance biometrics were the most sensitive for Urban>5%. Directly developed geodata was more successful than remotely sensed geodata in assessing coal mining impacts to habitat and biodiversity.

Chapter 4 – A GIS BASED WATERSHED RISK ASSESSMENT OF URBAN, AGRICULTURE AND COAL MINING IMPACTS UPON AQUATIC BIODIVERSITY

4.1 INTRODUCTION

Active surface and underground mines represent critical components of stream health assessment, along with agricultural activities. Owing to AMD's growing importance as a significant impact to ecology, the U.S. Environmental Protection Agency (USEPA) singled out acid drainage as the number one water quality problem affecting Appalachia. This research focused on quantitative assessment of aquatic biodiversity and associated ecological responses to active long-wall underground coal mining and abandoned underground and surface contour mining activities. Reclamation status of surface contour mines (e.g. strip mines) was also investigated in addition to mixed urban and agricultural activities. Both directly developed and remotely sensed land use and land cover information was evaluated.

Richards and Host (1994b) initiated a discussion of potential benefits that 1:250000 scale and finer geodata (e.g. 1:24000) can have in predicting landscape-biodiversity relations. Yoder and Rankin (1996) have also emphasized that biological assessments must be accompanied by appropriate characterizations of habitat quality and land use statistics. Both are needed to establish relationships between stressors that impact and degrade aquatic ecosystems, and resultant quality of the ecosystem. An evaluation was previously provided to assess relations in Leading Creek among land use/cover and aquatic habitat, where ASML and urbanization were found to significantly impact qualitative habit scores. This work compliments that analysis by examining landscape-biodiversity relationships directly.

4.1.1 Ecological risk Assessment Frameworks

Variation in several watershed parameters may compete and combine in the ultimate dominance of biological condition (Karr *et al*, 1981a, 1985; Minshall *et al*, 1985; Richards *et al*, 1994b, 1996). There are many subtle chemical and physical mechanisms potentially determining biological health (Chapman *et al*, 1992; Karr *et al*, 1981b; Rankin, 1989; Richards *et al*, 1994a, 1996; Rosenberg *et al*, 1986, 1993; USEPA, 1980, 1989). Research to date relates the challenge in using chemical and physical data to predict aquatic biodiversity on watershed, subwatershed, and finer scales (Allan, 1995; Bicknell *et al*, 1993; Cherry *et al*, 1995; Milhous, 1982; Richards *et al*, 1994b; Waters *et al*, 1995; Xia, 1997; Xinhao *et al*, 1997). The studies point out that physical, chemical, and biological data need to be more fully appreciated, and integrated in context, when defining management schemes to improve ecology (Cairns *et al*, 1991; NRC, 1992; Yoder *et al*, 1996). Physical, chemical, and biological monitoring programs complement each other (Miller *et al* 1988) and have determinant contributions to overall ecosystem health (Allan, 1995; Karr *et al*, 1981a, 1981b; Rankin, 1989). The role of land/use cover may prove to be a useful integrating tool.

Underlying processes and functions of the entwined physical, chemical, and biological system need to be investigated to accurately assess damaged ecology (NRC, 1992). Such studies must be conducted on scales that can incorporate critical features of the system, i.e. those that contribute significantly to variation in endpoints (or dependent variables) of interest (USEPA, 1996, 1998). For examination of freshwater aquatic ecology, this may often involve large watershed-scale studies that can address a wide array of inputs influencing stream conditions nearby and far downstream (Arnold *et al*, 1995; Cairns *et al*, 1996; Donigian *et al*, 1995; NRC, 1992). In relating landscape to biodiversity, a relatively undefined framework exists today for conducting ecological risk assessments, particularly for complex systems with multiple stressors (Munkittrick *et al*, 1998; Suter, 1998a; USEPA, 1998).

Initial stages are being set for conducting proper integrated ecological risk assessments, but there has remained a limited ability of these approaches to clearly identify dominant, controlling causes and effects of decline in aquatic ecology (Karr 1991; USEPA, 1996). Cause and effect relationships are desired but may not be readily discerned or identified through current methodologies. This will be more true particularly for simplified monitoring programs that might rely on a single endpoint (Chapman, 1995). This example and other efforts do indicate, however, that Geographical Information Systems (GIS) can be used to successfully assess landscape impacts to water quality and biodiversity (Osborne *et al*, 1988; Richards *et al*, 1994b). The relative complexity of the hydroecological system will continue to impede our ability to link sources of “nearby” stress directly to degraded ecology instream. Ecological stress imparted by man is often controlled by activities upstream, and in most cases due to activities on the landscape, and not in the channel itself (Dunne *et al*, 1978; Leopold *et al*, 1964; USDA, 1998). Investigation of finer scales of land use/cover data may lead us to a more productive framework for investigating field-scale phenomena. Scales of 1:24000 geodata sets were investigated here in assessing biodiversity.

4.1.2 Landscape-Biodiversity Study Objectives

To evaluate AMD within an integrated risk assessment framework, collection and comparison of spatial and temporal data describing four profiles of the watershed were eventually undertaken. Fundamental watershed traits chosen covered (a) hydrology-hydraulics, (b) chemistry of natural sediments and waters, (c) insect and fish community assemblage, and (d) lab and *in situ* toxicity test analyses. Supporting the investigation, fieldwork also included quantitative and qualitative instream and riparian habitat assessments (Anderson *et al*, 1976; Delong *et al*, 1991; Gallagher *et al*; 1999; Rankin, 1989; USEPA, 1989).

The hypothesis investigated here addressed aspects of hydrologic-based watershed-scale ecological risk assessment by employing GIS-based land use/cover analysis. The objective was to evaluate quantitative descriptions of agriculture, urban, surface and underground coal mining activities, and to relate their impacts directly to biodiversity. Active and abandoned mining were considered. Few active and abandoned mine studies exist that combine GIS at fine scales of 1:24000, land use/cover analysis, and ecological risk assessment tasks (Richards *et al*, 1994b, 1996). This paper integrated these approaches by testing the hypothesis that, at a scale of 1:24000, land use/cover variables describing urban, agricultural, and various mining activities can be used to characterize risk to aquatic biodiversity. The hypothesis was investigated by comparing land use/cover variables directly to measures of benthic macroinvertebrate data. Urbanization, AUML, and to a lessor degree ASML showed significant stresses to biodiversity where active mining and agriculture activities were relatively insignificant stressors.

4.2 STUDY WATERSHEDS

Leading Creek is a fifth order stream (scale: 1:24000) 96% of which spans Meigs County, with other portions in Athens, and Gallia Counties. Together, 11 mainstem stations and 19 tributary stations shown in Figure 3.1 in Chapter 3 were selected for hydrologic, hydraulic, and ecological risk assessment monitoring. Mainstem stations were located to provide complete longitudinal coverage of Leading Creek, and discrimination among potential sources of sediment or contaminant impacts to the stream system. Monitoring also included 3 reference sites outside of Leading Creek. Biological monitoring characterized the surrounding reach, and cross-section zones defined all other sediment, water quality, and hydrologic-hydraulic properties discussed.

4.2.1 Leading Creek Mainstem and Tributary Stations

Ten mainstem sites and six of the tributary sites were included in stream stage monitoring indicated by designation as staff gage stations in Figure 3.1 in Chapter 3. Agricultural activities dominate the upper mainstem while downstream reaches are dominated by mining. Eighteen tributary stations in the Leading Creek Watershed received some level of consistent sampling effort for most variables described here. These stations were initially chosen based on association with either (1) their known or suspected role in the contribution of point source pollutant loading to Leading Creek, or (2) their significance based on the size and nature of associated drainage area. The six tributaries selected for stream stage monitoring were chosen based on proximity to significant points on the mainstem of Leading Creek, proximity to known point sources and suspected nonpoint sources, or based on their significant representation of major tributary inflows. Parker Run (TS11) and Ogden Run (TS6) receive active coal mining treatment plant effluents and assorted stormwater and sanitary discharges. Other NPDES sites relate to sanitary treatment (STP), water treatment (WTP), and quarry discharges. Leading Creek's average fall is 0.16% (100*km/km) (OEPA, 1991b). Mainstem slopes of the stream bed range 1% to 0.01%, from headwaters at Albany, Ohio to the mouth of the Ohio River, respectively.

4.2.2 Ecoregion Reference Watersheds

Three comparative reference stream sites in two watersheds were selected on the basis of (1) exceptionally high IBI/ICI biodiversity scores in historic assessments, or (2) visibly exceptional habitat characteristics that are representative of what might be attainable in the Western Allegheny Plateau ecoregion. IBI refers to the site's Index of Biotic Integrity, a fish-based metric, and ICI refers to the Invertebrate Community Index, an insect-based metric (Karr, 1991; Miller *et al*, 1988; OEPA, 1989; USEPA, 1996). Upon initial inspection, the reference sites did not indicate marked decline but were potentially impacted by various anthropogenic traits including adverse effects from urban, agricultural, ASML, or AUML sources. Due to the wide reach of AML in Appalachia, there were no readily identifiable similar sized watersheds nearby that could claim strong independence from all or most of these stresses.

Federal Creek, a primary tributary of the Hocking River, spans Athens and Morgan Counties and drains 375 km^2 with mainstem length of 38 km. The subwatershed area monitored was 68 km^2 comprising the northwestern headwaters above Amesville, Ohio on State Route 329 at County Road 35. The average energy line fall of Federal Creek mainstem is 0.36%. Federal

Creek was noted by OEPA to have in-place contaminants (OEPA, 1991a) and is subject to AML-AMD influence based on review of U.S. Geological Survey (USGS) land use coverages (scale 1:24000). Symmes and Leading Creeks are both within Ohio EPA's study group of southeast tributaries draining to the Ohio River. Symmes Creek covers portions of Lawrence, Jackson and Gallia Counties, and is 113 km in length with slope 0.064% and drains an area of 920 km^2. The upper and lower reference sites each drain 348 km^2 and 897 km^2, respectively. The lower site is near State Route 243 below McKinney Creek, and the upper site near State Route 141, above Sand Fork. Upper Symmes Creek is more influenced by ASML and AUML than the lower site.

4.3 METHODS AND MATERIALS

Data collection in Leading Creek and the reference watersheds included monitoring of: (1) Land use and land cover, (2) precipitation, stream stage, water velocity, and flow; (3) sediment and water quality; (4) benthic macroinvertebrate and stream habitat; and (5) toxicity through sediment, acute, and *in situ* testing programs. To assess chemical, physical, and biological interactions, stations in Figure 3.1 were examined over a wide range of drainage areas and stream orders (Minshall *et al*, 1985; Dunne *et al*, 1978; Vannote *et al*, 1980; USEPA, 1996). Data collection methods for water quality indicators and biodiversity are detailed here. Water quality indicators were used to assess AMD impacts and to better examine trends in biodiversity.

4.3.1 Land Use and Land Cover

Spatial analysis of subsheds used several geodata sources including data hand-digitized from USGS quadrangle 7.5-minute map series, field data, and USGS hydrology geodata obtained from Ohio Division of Natural Resources (ODNR). Land use/cover data (scale 1:24000) was developed through four primary sources including a) 7.5-minute quadrangle series maps depicting abandoned strip mines, b) Ohio Division of Natural Resource (ODNR) geodata depicting abandoned underground mines, c) 1988 Landsat5 Thematic Mapper (TM) geodata developed by ODNR specifically for assessment of abandoned mine lands, and d) 1994 Landsat5 TM geodata developed by ODNR for general environmental assessment. Multi-spectral TM processing evaluates visible and infrared electromagnetic radiation from the earth's surface. Two Landsat5 Thematic Mapper data sets (resolution = 30x30m) describing watershed land use/cover characteristics were incorporated. Described in Chapter 3, definitions, processing and interpretation of Landsat5 data were provided by ODNR.

4.3.2 Insect and Fish Community Assemblage

Data collected in 1995 during planning for this study were amended with surveys of biota in 1996 and 1997 and habitat in 1996. Trained, experienced technicians conducted all biological and toxicological sampling and analysis. A contract ecological services laboratory familiar with Ohio streams, fish, and insects performed sample identifications. Various biological sample collections were carried out during appropriate spring and fall seasons. For biological data collection, stations represented nearby reach areas of the stream, incorporating typical periodic features and structure defined within the banks. In early August and late September 1996, using a combination of electroshocking and seining, fish community structure, age-class, and growth were assessed at sites along the mainstem and at the Federal Creek site. Riffle, pool, and shoreline habitats at each station were sampled intensely using D-framed nitex-mesh dipnets, and samples preserved in ethanol (OEPA, 1989). Qualitative dipnet samples were collected twice for tributary sites, and three times for mainstem sites from 1996 to 1997. Quantitative biodiversity sample collection was also attempted in spring and fall of 1995 and 1996 on mainstem sites using multiple-plate Hester Dendy samplers colonized for 6 weeks.

Dipnet samples were sieved (0.5 mm mesh) and rinsed in the laboratory to remove ethanol, placed in an enamel pan, and separated from fine substrate for enumeration and identification. Identifications were made with the aid of various taxonomic keys (Mason 1973; Pennak 1989; Merrit *et al*, 1984; Wiggins 1977). Taxonomic values derived included total taxon richness and abundance, mayfly abundance, Ephemeroptera-Plecoptera-Trichoptera (EPT) abundance, % EPT abundance, and EPT richness. Taxon richness is referred to here as taxa.

4.4 RESULTS

4.4.1 Land Use and Land Cover Analysis

Land use and land cover analysis addressing urban, agricultural and mining activities is summarized in Figures 3.2 through 3.7 in Chapter 3. Figures 3.2 and 3.3 depict agricultural activity. Figures 3.4 and 3.5 depict mining activity derived from remotely sensed data, and Figures 3.6 and 3.7 describe directly digitized data derived from either aerial photography (ASML) or permit data (AUML). Additional detail regarding land use/cover analysis results is found in Chapter 3 and Appendix A. This includes a summary of the history of coal mining in Leading Creek dating back to 1833, and background ecological risk assessments of the watershed conducted in the last decade. Assignment of Bare Soil as agricultural activity was previously found to be consistent with recent field surveys for all subsheds except headwaters stations TS1 and TS2 which drain portions of the Town of Albany, Ohio. June 1988 Landsat5 representations for TS1 and TS2 were well described by the urban coverage with possibly modest increases in urban density. The Urban (1988 TM) category was to be assigned without respect to this qualification on Bare Soil. Analysis here incorporated active mine operations with and without gob and spoil to address potential biases in classifications of Mine Gob (1988 TM), Mine Spoil (1988 TM), and Barren Lands (1994 TM).

4.4.1.1 Overview of Point Sources and Non-Point Sources

In comparison to active underground mining, minor point source permitted National Pollutant Discharge Elimination (NPDES) discharges given by the WTP, STP in Figure 3.1 in Chapter 3 and sanitary wastewaters from the active mines represent insignificant loading of treated effluents and flow into the stream system. This was expected assuming proper operation of standard treatment systems indicated to be in place. The active mining operations must meet categorical effluent limitations established for coal treatment plants. Treatment of coal preparation wastewater involves primarily physical separation processes and pH neutralization using sodium hydroxide. Waste sludge results from precipitates of primarily iron, manganese, and aluminum oxy-hydroxides and co-precipitation of heavy metals (Diz, 1997; USEPA, 1976).

Coal preparation plant wastewater effluent is high in total dissolved solids (TDS) primarily composed of sulfate, sodium, and chloride where conductivity at SOCCO Mine No.31

typically ranges between 4.0 to 6.2 mmhos/cm. During lower flow conditions the watershed can gain significant alkalinity from deep long wall mining mine operations, helping buffer AML-AMD entering the lower mainstem. The quarry discharge is also a significant contributor of headwater alkalinity during low and medium flows with average pH 7.9 at LCS1. For most periods of flow, Mine No.2 wastewaters are conveyed to Mine No. 31 for treatment and storage before discharge to Parker Run (TS11). Following periods of increased groundwater infiltration after run-off, some process wastewaters are treated and discharged to Ogden Run (TS6).

Included in the land use/cover category AUML>2% are tributary stations TS15, TS16, and TS17, and mainstem station LCS10 directly below TS17. Station TS15 lies above LCS8 and TS16 above station LCS9 but AMD loading from these tributaries are not typically discernable in mainstem water quality data. With average pH 6.7 and 7.3, respectively, TS15 and LCS10 are less impacted by AMD than stations TS16 and TS17. The latter stations constitute AUML>15% and have average pH 4.6 and 5.5, respectively. LCS10 is the only mainstem site with AUML>2%. Sites with severe AMD are distinguished here by AUML>2% in comparing biodiversity and land use/cover. Significant differences exist between sites impacted by severe AMD, Mine No. 31 with average pH 8.1, and other sites with pH 7.4 to 7.7.

Specific conductivity can also be used to highlight differences between AMD impacted reaches with moderate levels, Mine Nos. 2 and 31 and lower mainstem sites with highest levels, and decreasing levels from LCS1 in the upper mainstem attributed to discharges from a closed quarry operation. The limestone quarry above LCS1 is responsible for increased pH and conductivity observed at that station. While not directly studied, a small closed Town of Albany landfill is also located just upstream of the quarry influence. The scope of this paper was limited to examining sites with and without severe AMD impact. A proportional relationship between lower pH and poor biodiversity has been established for Leading Creek and other streams subject to stream acidification processes (Allan, 1995; Gallagher *et al*, 1999; USEPA, 1980).

4.4.2 Benthic Macroinvertebrate Biodiversity

Spring, summer and fall reach results (riffle+run+pool) and site averages for qualitative dipnet collections of benthic macroinvertebrate are given in Figure 4.1 and Table B1 in Appendix B. Tributary and mainstem sites were sampled twice in 1996 (EPT 2x); Parker Run and mainstem sites three times in 1997 (EPT 2x); and all sites below Mine 31 twice in 1995 (EPT 0x). The curves developed for EPT Taxa Richness and Total Taxa Richness in Figure 4.1 indicate that associated abundance is strongly related to taxa; by close approximation a square root transformation for EPT and root 2/3 for total insect family richness.

The TS16 data point omitted from curve development in Figure 4.1 points out how preferential selection of a few insect species can occur in severely stressed environments. Low taxa in acidified streams can still exhibit robust abundance. In spring 1996, only 2 species in the EPT index were collected at Titus Run and none were found at stations Paulins Hill Run (TS16) and Thomas Fork (TS17). For fall collections the same year, only 2, 1, and 2 EPT species, respectively, were found at these sites. Odonata (Dragonflies), Ephemeroptera (Mayflies), and Plecoptera are aquatic insects severely affected by acid mine drainage where species tolerant of

AMD may also include several Diptera, species of Chironomidae, and at least one Megaloptera, Sialis sp. (Cherry *et al*, 1999; Herricks *et al*,1974).

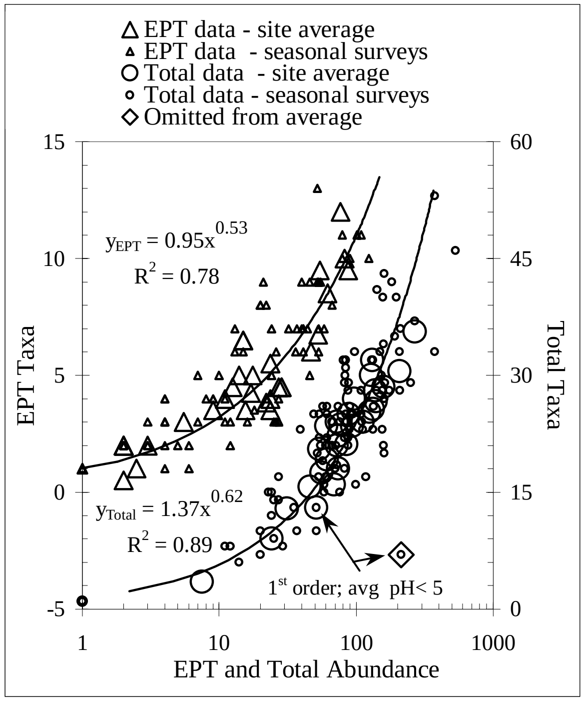

Figure 4.1 – Leading Creek Qualitative Benthic Macroinvertebrate Results – 1995 to 1997

Qualitative data was used here to extend evaluations to a wide range of drainage areas and stream orders otherwise not possible with quantitative data. Quantitative Hester Dendy data derived from artificial substrate samplers typically required a large drainage area (1000 ha+) to achieve adequate flow requirements and was difficult to collect in stream reaches with significant scour and deposition of sediment. Leading Creek quantitative macroinvertebrate sampling was attempted only on the mainstem and was often compromised or invalidated by these two factors where flow was sometimes insufficient or sedimentation buried the samplers. Quantitative unit area data is highly desirable, but may not typically be amenable to subwatershed scale risk assessments. Qualitative dipnet sampling performed in accordance with State of Ohio procedures (OEPA, 1989) does at least attempt to reduce bias in estimation of taxa richness and abundance by rigorous sampling of all nearby riffle, run, and pool zones.

Lower Symmes Creek biodiversity data was taken from the 1990 assessment by OEPA where a designation of exceptional warmwater habitat (EWH) criteria was given for a nearby reach 3 km downstream of (R)SCS2 (OEPA, 1991b). Qualitative dipnet collections at Leading Creek station LCS7 for the same time frame gave 21 total taxa and 4 EPT taxa, respectively, and a designation of good warmwater habitat criteria. In 1991 OEPA again assigned the EWH designation to the upper section of Federal Creek (above Sharp's Fork) in their surveys of 1990 (OEPA, 1991a). Previously EWH attainment status was also assigned in the 1978 Ohio WQS (Water Quality Standards). These evaluations were based on observations of habitat and water quality, and biodiversity encompassed in ICI and IBI indexes.

Examining length class histograms, no discernable trends in mainstem fish growth data were noted for sample collections between early August and late September 1996 using seining and shocking collection methods. This was based on species available at each site where sufficient samples were not available at LCS9 and LCS10 (Cherry *et al*, 1999). Only limited fish data were available for the study and included historical and current IBI data for some mainstem sites. Historic IBI data was primarily available for middle and lower mainstem reaches where a drop in total scoring was observed moving from values of 38 and 42 just above Parker Run down to a level of 25 at LCS10. ICI data extending from stations LCS1 to LCS10 had spatially mixed results with average 32.3 (σ =7.8, n = 9) noting LCS9 was invalidated by excessive sedimentation from AML. More widely available qualitative biometrics of insect taxa and abundance were expectedly weakly representative of ICI data in larger subsheds of Leading Creek. Quantitative EPT taxa was the most strongly correlated to ICI.

4.4.3 Landscape-Biodiversity Relationships

The relationship between land use and land cover and biodiversity was pursued along analogous lines investigated for habitat data. Spearman-rank data is given in Table 4.1. Similar features were noted for correlation of landscape-biodiversity relationships as were noted for landscape-habitat relationships. Deleterious land uses (-) were all associated between total insect data and mining or 1988-1994 woodlands data (r_s = -0.46 to -0.69). Again parameters representing agricultural land uses or at least not woodlands were associated with positive influences (r_s = 0.40 to 0.69). In landscape-biodiversity correlations, total taxa data was far more significant for more landscape parameters than was EPT data. In the case of EPT data, community strength appeared to be more consistently, but still weakly correlated to AML land uses except for EPT abundance that was also weakly correlated for AUML. EPT taxa was also weakly (-) correlated for woodlands where Ag/Urban was weakly (+) correlated (1994 TM).

Figures 4.2a and 4.2b also delineate specific relationships for %AUML>0 and %ASML>0, where both land use categories were found to be the consistently significant with respect to loss of biodiversity. In Figures 4.2a four sites had levels of ASML activity with AUML = 0%. These included Lasher Run (TS13; 11%), Grass Run (TS10; 5.3%), and upper and lower stations on Mud Fork (TS8A; 5.5% and TS8B; 3.6%, respectively). These sites showed typically no significant response in AMD parameters compared to severely impacted Thomas Fork (TS17) with ASML=6.6% and AUML=15.9%. Little Leading Creek (TS14) also showed little AMD impact with ASML=8.5% and AUML = 1.7%. Titus Run at ASML=15% and AUML =2.7% showed only borderline AMD with average pH 6.7, where it was grouped with AUML>2%. Shown in Table 4.1 as %URASML indicating unreclaimed ASML lands, no differences were noted between ASML and unreclaimed ASML. The data and correlations in Figure 4.2a and 4.2b differ from those in Table 4.1 which included sites with ASML=0 and AUML=0. For unreclaimed ASML, slightly poorer results were obtained (R^2 = 0.33 to 0.32).

The functional relationships for AUML given in Figure 4.2b were both significant with acceptable test assumptions (total taxa: n = 9, [P = 0.0017]; α = 0.05, 1-β = 0.97; EPT taxa n = 9, [P = 0.0015]; α = 0.05, 1-β = 0.99). Differences emphasized in correlations in Figures 4.2a and 4.2b indicated that AUML>2%, and not ASML>3%, was a more reliable threshold associated with poor total taxa and poor EPT taxa. Figures 4.2a and 4.2b also included non-zero reference site biological data available at stations (R)FCS1 and (R)SCS2 for both ASML and AUML.

Table 4.1 – Spearman-Rank Correlation Coefficients for Land Use/Cover and Biodiversity;
(Blank data indicates significance level [P] >0.05; n = 28-30)

Land Use/Cover Category	Total			EPT		
	Taxa	Abundance	Community Strength	Taxa	Abundance	Community Strength
Directly Digitized LUC						
%ASML	-0.59	-0.46	-0.60	-	-	-0.40
%URASML	-0.57	-0.47	-0.61	-	-	-0.41
%AUML	-0.47	-	-0.49	-	-0.39	-0.38
%UAML+%URASML	-0.59	-0.48	-0.61	-	-	-0.42
Below Mine No. 31	-	-	-	-	-	-
1988 Landsat5						
%Mine Gob	-	-	-	-	-	-
%Mine Spoil	-	-	-	-	-	-
%Gob (No SOCCO)	-	-	-	-	-	-
%Spoil (No SOCCO)	-	-	-	-	-	-
%Bare Soil	0.64	0.56	0.65	-	-	-
%Sparse Vegetation	0.56	0.58	0.60	-	-	-
%Mod. Vegetation	0.62	0.54	0.63	-	-	-
%Full Vegetation	0.48	-	0.42	-	-	-
%Shrub	0.50	-	0.46	-	-	-
%Wooded	-0.68	-0.54	-0.66	-	-	-
%Water	-	-	-	-	-	-
%Wetland	-	-	-	-	-	-
%Urban	-	0.39	-	-	-	-
1994 Landsat5						
%Urban	-	-	-	-	-	-
%Ag/Urban	0.69	0.53	0.66	0.48	-	0.44
%Shrub	0.40		0.43	-	-	-
%Wooded	-0.69	-0.58	-0.69	-0.40	-	-
%Water	-	-	-	-	-	-
%Wetland	-	-	-	-	-	-
%Barren	-	-	-	-	-	-

Tables 4.2 and 4.3 and Figures 4.3 and 4.4 describe results of hypothesis tests conducted to test the influences of various thresholds of land use/cover upon total macroinvertebrate and EPT taxon richness and abundance. Figures 4.3a and 4.4a describe total and EPT abundance results, respectively and Figures 4.3b and 4.4b show total and EPT taxa, respectively. Because of non-normality of residuals associated with taxa and abundance Kruskal-Wallis one-way ANOVA ranks were performed in some cases, using data corrected for repeated values. For non-parametric ANOVA, the Kruskal-Wallis multiple comparison z-value test was used to evaluate significant results employing the full Bonferroni criteria for all groups directly compared (Hintze, 1997; Zar, 1999). The power for results stated in Tables 4.2 and 4.3 are given for parametric ANOVA results.

For Table 4.2 power (1-β) ranged 0.23 to 0.55. In Table 4.3 power ranged 0.97 to 1.00 except for total abundance at 0.86.

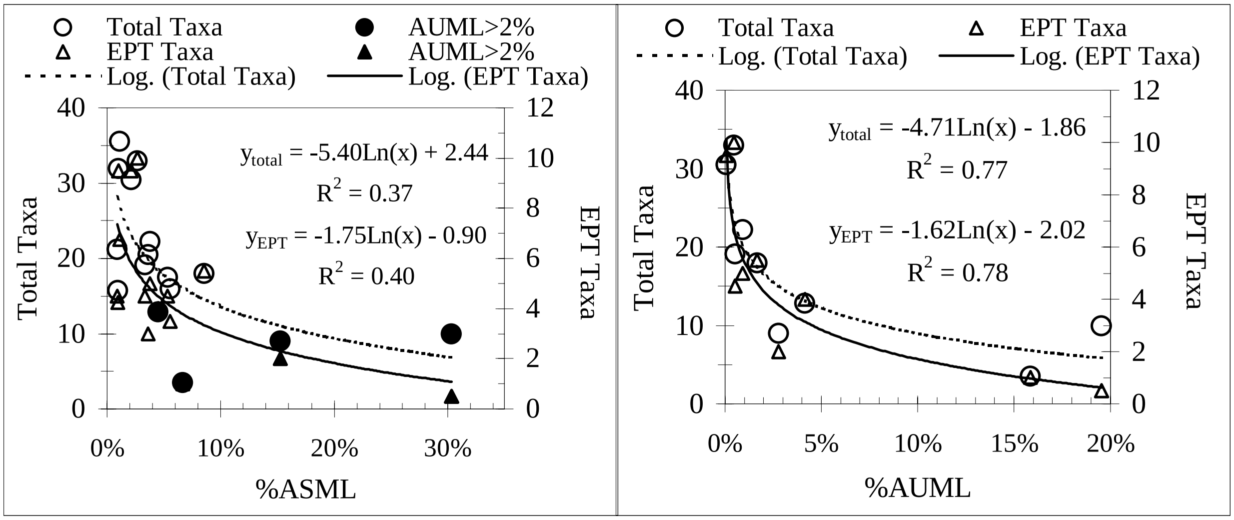

Figure 4.2a and 4.2b – Biodiversity versus (a) %ASML>0 and (b) %AUML>0

Table 4.2 – ANOVA Results for Land Use/Cover-Biodiversity Relationships - Minor Strip Mining and Agricultural Emphasis; (No comparisons were significant)

Group #	Land Use/Cover Category	# Sites	Average Total		Average EPT		Range of Drainage Areas (km^2)
			Taxa	Abundance	Taxa	Abundance	
1	Reference Sites (R)FCS1; (R)SCS2 taxa only	1-2	32.0	208	10.0	88.0	67.9 - 896
2	Active Mines	2	28.5	148	5.0	34.5	18.9 - 14.8
3	ASML = 0 *(w/ Urban < 5%)*	9	25.1	104	7.0	36.7	6.1 - 86.1
4	0 <ASML< 3%	4	26.3	132	6.5	38.0	161 - 209
All	Parametric Test ([P]; α = 0.05)	16-17	Non- 0.21	Non- 0.20	Yes 0.30	Yes 0.33	1.2 - 387
3	Bare Soil >3% *(w/ Urban <5%)*	6	26.0	119	8.2	43.3	9.7 - 86.1
4	BareSoil <3% *(w/ ASML<3%)*	7	25.0	107	5.7	31.9	6.1 - 209
All	Parametric Test ([P]; α = 0.05)	16-17	Yes 0.35	Non- 0.15	Yes 0.075	Non- 0.45	1.2 - 387

Because of reduced sensitivity inherent in the non-parametric multiple comparisons, the biodiversity-land use/cover experiment first examined whether or not differences could be discerned between weakly influential AML, non-AML, and active mining. This was guided by lack of habitat and water quality degradation associated with active underground mines and agricultural land uses with ASML<3%, AUML=0, and Urban<5%. With no significant differences in biodiversity realized among mining activities in this population of sites, a second comparison was performed evaluating the significance of agricultural activities looking again at a cut-off of Bare Soil=3%. Shown in Table 4.2, results indicated no significant differences between active mining, ASML<3% and AUML=0, or between agricultural activities with %Bare Soil data above or below 3%. Evaluations were performed with similar results examining AML and agricultural influences without distinguishing TS6 and TS11 as active underground mine (Active UM) tributaries. Not presented in Table 4.2, all results were also insignificant for community strength comparisons.

Table 4.3 – ANOVA Results for Land Use/Cover-Biodiversity Relationships - All Groups;
(For significant tests, significant differences between groups are identified by group #'s in {})

Group #	Land Use/Cover Category	# Sites	Average Total		Average EPT		Range of Drainage Areas (km^2)
			Taxa	Abundance	Taxa	Abundance	
1	Reference Sites (R)FCS1; (R)SCS2 taxa only	1-2	32 {4,5}	208 -	10 {2,5}	88.0 {All}	67.9 - 896
2	Urban >5%	2	25.5 {5}	95.5 -	2.0 {1}	2.0 {1}	1.2 - 1.7
3	ASML <3%	15	25.9 {4,5}	117 -	6.6 {5}	36.8 {1,5}	6.1 - 209
4	ASML>3% *(w/ AUML < 2%)*	7	19.7 {1,3,5}	66.0 -	4.6 -	18.0 {1}	4.6 - 304
5	AUML>2%	4	9.0 {All}	49.0 -	2.0 {1,3}	4.8 {1,3}	1.6 - 387
All	Parametric Test ([P]; α = 0.05)	29-30	Yes $1x10^{-6}$ *	Non- 0.018*	Non- 0.0018*	Yes; qual. 0.0012*	1.2 - 387

4.4.3.1 Less Dominant Mining and Agricultural Activities

Along similar lines discussed in Chapter 3 to select Bare Soil=3% as a threshold criteria for agriculture, evaluation of 1994 Landsat5 Ag/Urban data resulted in selection of a similar set of sites with a separation value of Ag/Urban=25% to 26%. Major differences against Bare Soil data partitioning were that many sites were close to the Ag/Urban threshold, and Five Mile Run (TS3) was assigned to low activity. Five Mile Run was identified in reconnaissance as one of the most heavily impacted sites with only agricultural influences. Based on Ag/Urban data and its association with mixed urban activity, poor discrimination in magnitude at non-urban sites, and time frame of remote sensing in fall, Bare Soil data was retained as a final discriminator for agricultural influences. This also allowed consistent use of a single Landsat5 image for final comparisons. This choice may be unimportant given lack of significance observed with biodiversity and Bare Soil, and similar correlations between these variables and biodiversity, shown in Table 4.1. Roth who studied 23 subsheds in Michigan showed an agriculture level >60% was needed to see significant differences in total habitat scores (Allan, 1995; Roth, 1994). For 1994 Landsat5 Ag/Urban data, only TS1 at 81% met this criteria. Bare Soil data was found in other work to be highly correlated to TSS load rates, supporting its selection. For Tables 4.2, and 4.3, qualification of group membership is included for strict interpretation of land use/cover categories.

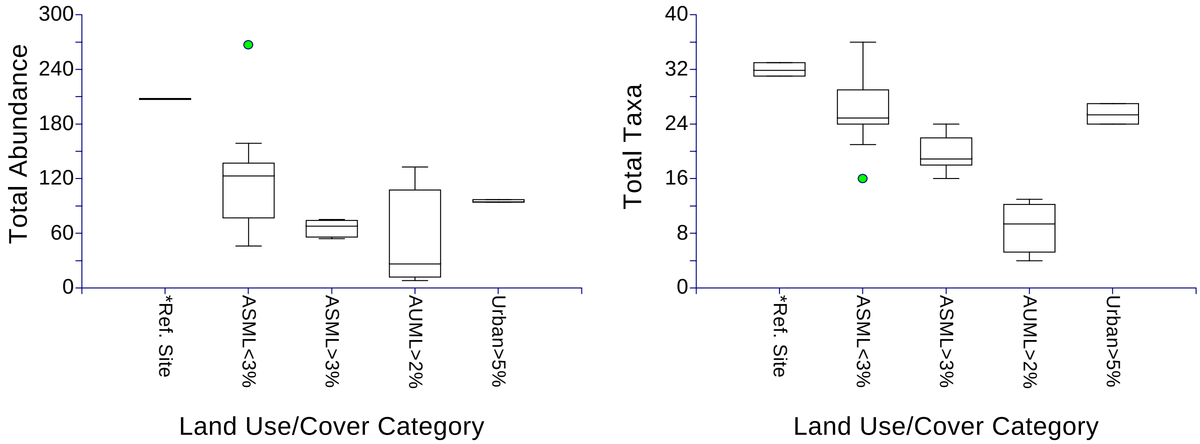

Figure 4.3a and 4.3b – Land Use/Cover Comparisons for Total Benthic Macroinvertebrate (a) Abundance and (b) Taxa Richness

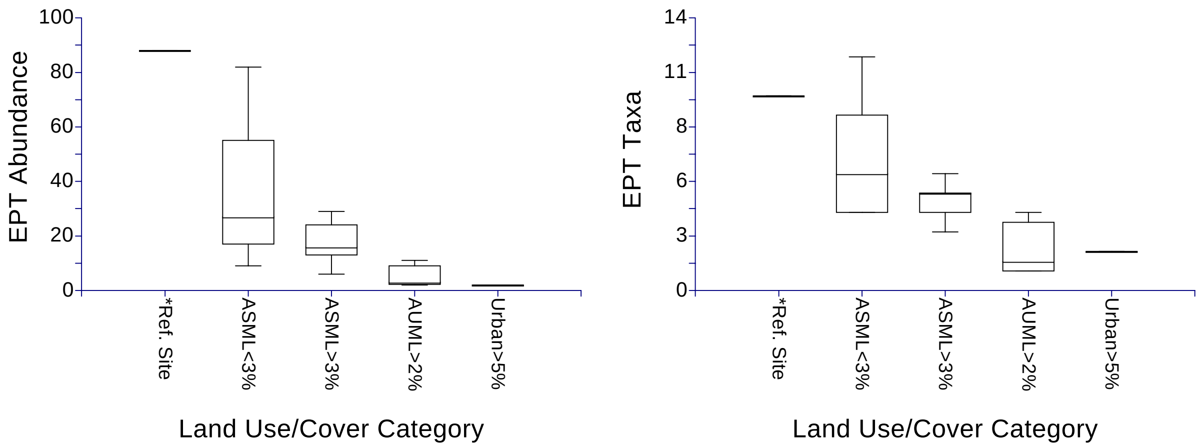

Figure 4.4a and 4.4b – Land Use/Cover Comparisons for EPT Benthic Macroinvertebrate (a) Abundance and (b) Taxa Richness

4.4.3.2 Comparisons Between Urban, ASML, and AUML Land Uses

Indicated in Figure 4.3b and Table 4.3, with respect to the average reference site total taxa (32) data and ASML<3% (25.5), significant differences were noted between average total taxa for land use/cover categories ASML>3% (19.7) and AUML>2% (9). Total taxa for sites with ASML>3% was also found to be significantly different from sites with ASML<3% (25.9). No significant differences in average total taxa were observed between reference sites and sites with Urban>5% or AMSL<3%. AUML>2% was significantly lower than all sites, including ASML>3%, ASML<3%, and Urban>5%. Results for landscape-biodiversity comparisons are further summarized in Table 4.4. This table presents the most significant result for parametric and non-parametric tests. Qualified results indicate that a rejected assumption for normality or equal variance for a parametric comparison was borderline with respect to the significance level (α = 0.05), or that full inter-group non-parametric test results could not distinguish differences between the two groups with inclusion of sites with ASML<3%.

Table 4.4 –Multiple Comparison Test Results for Land Use/Cover-Biodiversity Relationships;
((=) – no significant difference, (<>) – significant difference, (<>p) – significant difference qualified by borderline rejection of some parametric test assumption, (<>o) – significant difference qualified by omission of group ASML<3%)

Land Use/Cover Category	Average Data				Urban >5%				ASML <3%				ASML >3%				AUML >2%			
	Total		EPT		Total		EPT		Total		EPT		Total		EPT		Total		EPT	
	Taxa	Abun.	Taxa	Abun.	Taxa	Abun.	Taxa	Abun.	Taxa	Abun.	Taxa	Abun.	Taxa	Abun.	Taxa	Abun.	Taxa	Abun.	Taxa	Abun.
Urban >5%	26	96	2.0	2.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
ASML <3%	26	117	6.6	37	=	=	<>p	=	-	-	-	-	-	-	-	-	-	-	-	-
ASML >3%	20	66	4.6	18	=	=	=	<>o	<>	=	=	=	-	-	-	-	-	-	-	-
AUML >2%	9.0	49	2.0	4.8	<>	=	=	=	<>	=	<>	<>p	<>	=	=	<>o	-	-	-	-
Ref. Site(s)	32	208	10	88	=	<>o	<>	<>p	=	=	=	<>p	<>	<>o	<>op	<>p	<>	<>o	<>	<>p

For full inter-group comparisons on total abundance data in Figure 4.3a, the non-parametric test rejected the null hypothesis that all medians were equal, indicating that at least two medians were different. The multiple comparison Kruskal-Wallis tests though could not distinguish which groups were significantly different due to significant variation among sites with ASML<3%. An outlier identified at LCS7 was attributed to explosive abundance in spring and fall 1995 and spring 1996 with values of 529, 376, and 376, respectively, and for later events falling to more frequently encountered levels at other sites with exceptional habitat. An additional parametric comparison excluding the group for ASML<3% (n = 14, [P = 0.0075]; α = 0.05, 1-β = 0.90) indicated significant differences between reference site total abundance (208) and Urban>5% (96), ASML>3% (66), and AUML>2% (49) land use/cover categories.

An exception with respect to use of parametric results for Table 4.3 was made for EPT abundance data shown in Figure 4.4a. Residual skewness normality was rejected at a significance level of 0.05 ([P}=0.044). AUML>2% though could not be distinguished from reference site EPT abundance data in the non-parametric multiple comparison, falling just outside of the interval of significance for the full Bonferroni criteria. The more powerful full inter-group parametric results were retained with this qualification, noting residuals passed kurtosis, omnibus, and homogeneity of variance tests. The qualified test indicated significant differences between all groups and reference data (88), where ASML<3% (36.8) and AUML>2% (4.8) were also significantly different. An additional unqualified parametric comparison excluding the group for ASML<3% (n = 14, [P = $3x10^{-6}$]; α = 0.05, 1-β = 1.00) was also investigated. Like the full inter-group parametric tests, results indicated significant differences between the average reference site EPT abundance (88) and ASML>3% (18), AUML>2% (4.8), and Urban>5% (2), where the latter two categories were different from ASML>3%. The qualified full inter-group test indicated similar results but did not distinguish significant differences between ASML>3% and the groups Urban>5% and AUML>2%.

Total taxa at sites with Urban>5% was not significantly different from mixed agricultural/urban/active mining entities in ASML<3%, but EPT taxa in Figure 4.4b showed significantly lower results. Recall that Urban>5% was also identified as having significantly poorer substrate quality than all other sites, and poor QHEI scores compared to reference data that were similar in magnitude to sites with ASML>3% and AUML>2%. For EPT taxa non-parametric methods were needed. Compared to reference data (10), significant differences in median EPT taxa values were found for both groups AUML>2% (2) and Urban>5% (2). ASML<3% was also identified with significantly higher EPT taxa compared to AUML>2% where ASML<3% (4.6) and ASML>3% (6.6) were comparable to the reference site. Not presented in Table 4.2, parametric null hypothesis tests (n = 30, [P = 0.00023]; α = 0.05, 1-β = 0.99) for EPT taxa were rejected for skewness normality ([P]=0.037), omnibus normality ([P]=0.045), and equal variance ([P]=0.0496) tests (α = 0.05). Significant results for non-parametric tests were recorded except multiple comparisons did not identify Urban>5% as different from ASML<3% or ASML>3% as being different from reference data, which the Tukey-Kramer identified. Without consideration of ASML<3%, ASML>3% was also found to be significantly different from reference data.

In Table 4.3 all results were significant for community strength for both EPT and total data. Due to lack of normality, non-parametric testing was necessary and multiple comparisons gave poor results for groups with low n including reference data and Urban>5%. For both EPT and total metrics, no group medians were identified as being significantly different from the reference site. For total community strength ASML<3% (249) was identified as being significantly higher than ASML>3% (153) and AUML>2% (76.6). For EPT community strength, significant differences were noted only between ASML<3% (83.2) and AUML>2% (9.9). These results appeared to strike a balance between findings for individual taxa and abundance parameters.

4.5 DISCSUSSION

Landscape-biodiversity relationships were investigated. Ideal for study of coal mining and AMD mechanisms in a watershed setting, anthropogenic impact in Leading Creek is segregated, becoming increasingly complex downstream with the addition of different sources of pollutants. Pollutant sources include mixed agricultural and urban influences, the heaviest of which occur upstream. Active underground mine treated effluents enter midstream, where mild historical AML and untreated AMD impacts also join, becoming more severe downstream. By comparing individual and combined impacts from mining operations against areas not mined, mining, agricultural and urban activities were analyzed for differences in aquatic biodiversity. Low pH and AMD impact established a subgroup of sites, TS15, TS16, TS17, and LCS10 with significantly different instream character, identified by AUML>2%. Strong relationships were also established between insect abundance and taxon richness for both total and EPT metrics.

To place biodiversity and land use/cover into context, similar to the work of Richards and Host (1994b), the first task undertaken was to understand habitat-biodiversity and landscape-habitat relationships, outlined in Chapter 3. Biodiversity data was then directly evaluated for its relationship to landscape by comparing taxa and abundance biometrics of macroinvertebrate community assemblage to land use/cover variables at scales of 1:24000.

4.5.1 Biodiversity and Selected Biocriteria

There are various potential approaches to selection of biocriteria within a detailed examination of landscape-biodiversity relationships. Biocriteria based on fish and benthic macroinvertebrate are often used as biological end-points for stream restoration because they play an important functional role in stream ecosystems and can be effective as monitors of environmental conditions (Burton, 1992; Karr, 1991; Rosenberg *et al* 1986, 1993; USEPA, 1996). Specific evaluations of insect assemblage as suitable biocriteria in watershed risk assessment of time and space variant stress have also been demonstrated (Richards *et al*, 1992, 1994a, 1994b; Sweeney, 1993; Wright *et al*, 1984). Based on other researcher's efforts, for comparisons to land use/cover, total and EPT taxa data were expected to best represent watershed trends in biodiversity (1996c).

Taxon richness is typically inversely related to the degree of stress, whereas biotic indices attempt to summarize information on the tolerance of the macroinvertebrate community (Cherry *et al*, 1999; Lenat, 1993; USEPA, 1996). The North Carolina (NC) Division of Environmental Management also uses taxon richness of the most intolerant invertebrate groups (Ephemeroptera, Plecoptera, Trichoptera, or EPT) and a biotic index similar to that of Hilsenhoff's biotic index designed to detect the influence of oxygen demanding wastes on biota (Cherry *et al*, 1999; Hilsenhoff, 1987; NCDHNR, 1990; USEPA, 1996). Similar biotic indexes have also been created in earlier decades attempting to relate sedimentation impacts (USEPA, 1996).

This research focused on qualitative macroinvertebrate community descriptions that contribute in part to ICI (insect) scores and that would be expected to only partially support trends in ICI and IBI. IBI and ICI scores often are employed to determine stream use-attainment standards such as defining warmwater habitat criteria or exceptional warmwater habitat criteria in Ohio (USEPA, 1996). The State of Ohio is somewhat unique in its combination of the IBI and ICI scores to describe stream use-attainment. Due to the variable nature of water flow and sedimentation at proposed study sites, qualitative dipnet data was used over more sparse Hester Dendy quantitative insect data. ICI scores are based mostly on quantitative benthic macroinvertebrate sampling. Together with fish-based IBI scores, full evaluation of biotic integrity is difficult to complete across various sized subsheds, by and large limiting validated investigations to fairly well defined mainstem reaches of order 3 or higher (scale 1:24000). Simplification in describing biodiversity for comparison of non-biological metrics such as water quality and land use/cover is highly desirable. A critical issue is that simplification is not fully amenable to unbiased representation of overall biotic integrity, where the latter concept has moved steadily to the front of ecological based risk assessments (Karr, 1991; USEPA, 1996). How to best balance investigations of landscape-biodiversity relations will remain until we better understand key underlying processes and mechanisms that lead to measurable degradation.

Richards and Host's (1994b) approach examined detailed functional metrics of macroinvertebrate biodiversity in addition to total taxon and EPT taxon richness examined here. For example they looked at filters and tied agriculture to embeddedness. Our approach also looked at total and EPT insect taxa and abundance as biotic endpoints. Qualitatively sampled EPT taxa is normally also used to augment the otherwise quantitative ICI index that includes 9 metrics of total taxon richness and various species-specific taxa. The small select set of broader based biodiversity metrics chosen was more easily managed with an expanded list of land use/cover variables. All landscape variables were assessed as percentage of catchment area.

Ecologists will continue to favor incorporation of several metrics like richness and abundance when reflecting upon overall impact or assessed risk due to specific stress. To partially evaluate the viability of this trend in risk assessment programs, this work also attempted to interpret quantitative land use variables in light of evident natural functionality between taxon richness and abundance in Figure 4.1. A metric of overall community strength was defined by

$S_{Community} = Taxa^a + Abundance$; with a = 2 for EPT data, and a = 1.5 for total data. Raw data distributions of community strength may be found in Table B1 in Appendix B along with average community strength data examined here. Community strength as defined would be expected to mask preferential selection of species against proliferation. At the same time it provides a single endpoint for comparing both reactions to stress against a presumably balanced reference site. Less true for EPT data, most Leading Creek sites were functionally balanced with respect to community strength except at site TS16. Significantly different results comparing either taxa or abundance to community strength would not be expected in comparisons involving the full sample population. Only marginal improvement in correlation between habitat in Chapter 3 and biodiversity studied here was recognized through use of the constructed community strength indices that combined abundance and power-function transformed taxa.

4.5.2 Landscape-Biodiversity Relationships

Among a mixture of agricultural and urban land uses commingled with strip mines and active and abandoned underground mines, this work showed total taxa was a better indicator of degree of AML stress than was EPT taxa. Similar to conclusions that strip mines have significantly degraded aquatic habitat, ASML was also implicated in reducing macroinvertebrate taxon richness. In comparison, abandoned underground mines above stream drainage elevation were found to be more deleterious stresses to aquatic life than ASML, requiring only 2% of the aereal landscape to impart significant degradation. Biodiversity was decimated for sites with 15% to 20% AUML. Conditions of 3% < ASML < 11% imparted relatively minor effects when AUML<2%. The degree of maximum strip mining activity needed to decimate biodiversity could not be discerned due to commingled AUML and ASML when ASML was >11%. The nature of ASML and associated lack of impact to water quality was reconciled by evaluating both AUML and ASML jointly through quantitative, directly derived geodata at scale 1:24000. For remotely sensed geodata, Mine Gob, Mine Spoil, and Barren data from Landsat5 1988 and 1994 processed images were insignificant and ineffective in differentiating landscape-habitat and landscape-biodiversity relationships.

Summarized in Figures 4.3a, 4.3b, and Table 4.4, overall results for total taxa data were more discriminating than total abundance data in assigning significant impact from land use/cover categories ASML>3% and AUML>2%. For total taxa, AUML>2% was more severe upon biota than ASML>3%. For total taxa Urban>5% and ASML<3% on the other hand were not distinguishable from reference data where they were for total abundance. Compared to reference data, total abundance was more equally disturbed among land uses defined by ASML>3% and AUML>2%, including similar levels of degradation for Urban>5%. For EPT macroinvertebrate, total abundance also identified significant degradation for land uses ASML>3%, AUML>2%, and Urban>5%, where the latter two were similar but significantly more severe than ASML>3%. EPT taxa showed significantly lower results for AUML>2% and Urban>5% compared to other groups where, unlike total taxa, ASML<3% was not significantly higher than ASML>3%, but was still significantly higher than AUML>2%. A distinction between total and EPT taxa was that total taxa was comparable to reference data, but EPT taxa was significantly lower for Urban>5%. Limited in this study to 2nd order subsheds (scale 1:24000), Urban>5% exhibited extreme sensitivity to EPT taxa similar to effects of AUML>2%. This was consistent with other biological monitoring results in small urban Ohio watersheds that showed lower biological scores with an increasing degree of urbanization (Yoder *et al*, 1996).

No differences were noted between ASML and unreclaimed ASML. In landscape-biodiversity correlations total taxa was more significantly related to more remotely sensed landscape parameters than abundance data. In the case of EPT data and components of taxa and abundance, community strength appeared to be more consistently correlated to AML land uses except for EPT abundance that was weakly correlated for AUML. No significant differences in biodiversity were realized among mining activities in the population of sites with ASML<3%, evaluating criteria for active mines and ASML=0%. Results also indicated no significant differences existed between active mining and agricultural activities with Bare Soil data above or below 3% aereal extent. Overall results for macroinvertebrate analysis were most compelling in their ability to distinguish significant impact from ASML and AUML. Total taxa data at heavily urbanized sites was not significantly different from mixed agricultural, urban, active-mining, entities representing ASML<3% and ASML>3%, or reference site data, but EPT taxa showed significantly lower results compared to reference site data and ASML<3%.

4.5.3 Investigation of Landscape-Biodiversity Interactions

An increased focus on water quality and quantity will continue to align with increasing needs to quantitatively relate chemical and physical watershed variables (Bicknell *et al*, 1993, Donigian *et al*, 1995; USEPA, 1991, 1996; Xinhao *et al*, 1997). Added to this will be a desire to directly link hydrology and landscape to ecological biocriteria like taxa, abundance, and more complicated indices like IBI and ICI scores (Miller *et al*, 1988; Richards *et al*, 1992, 1994b; USEPA, 1996), despite limitations inherent in those attempts (Yoder *et al*, 1996). This study has shown that simple biodiversity metrics like total taxa and EPT taxa, both known to be proportionality related to stress imparted by stream acidification processes, can be successfully related to land use/cover variables normalized to drainage area (Lenat, 1993; USEPA, 1980).

As Karr points out (1986), species richness and composition can vary substantially with stream size and region. Without quantitative approaches that address spatial dependence, aquatic health assessment methods may lack objectivity. A concern may exist that spatial dependence effects in ICI and IBI data can easily be lost in standardized reporting or ignored during assessment and generation of subscores or the final site IBI score. Hydrology, geology, fluvial geomorphology, and pollutant input can also have significant effects on community assemblage both at regional scales (Hughes *et al*, 1994; Richards *et al*, 1996; USEPA, 1980) and more localized scales (Richards *et al*, 1994b). A major benefit in area-normalized landscape analysis is that the level of input stress to the stream system is proportionately integrated. Spatial dependency of underlying hydrologic-hydraulic driven mechanisms is quite important to water quality degradation processes such as AMD. This was shown 20 years ago by EPA where lower order watersheds (up to 3^{rd} and 4^{th} order for a comparative scale of 1:24000) appeared to have a significantly higher risk of acidification (USEPA, 1980). Increased risk was attributed to regional geology and buffering capacity notwithstanding quantitative information about magnitude of aereal disturbance.

Three points were made recently by EPA concerning use of components of biological criteria (USEAP, 1996). First, success in protecting biological integrity depends on the development of measurement endpoints that are highly correlated with assessment endpoints. Secondly, efforts to assess these endpoints must be broadly based where this facet ranks higher than the actual choice of specific biocriteria used (assuming they are holistic and well defined). Finally, the best approach to assessing biological integrity is an integrative one that combines assessment of the extent to which either elements or processes of integrity are altered. (USEPA, 1996). There is expanding use of biometrics to cover wider geographic areas (Oberdorff *et al*, 1992; OEPA, 1989, 1991a, 1991b; Chapman, 1995). Landscape-biodiversity linkage in Leading Creek indicate that efforts like the Ohio EPA's continued move to examine benthic macroinvertebrate within a context of biotic integrity can be successful, along with similar efforts by states like Nebraska (Bazata, 1991; OEPA, 1989; USEPA, 1989, 1996).

Ohio is one of the states farthest along in these types of efforts coinciding well with its advanced and active GIS community. Geodata can be quickly accessed today from freely available government maps ranging in scale from 1:24000 to 1: 2,500,000. These maps have several useful applications, particularly at watershed to regional scales (Richards *et al*, 1994b, 1996; Seaber *et al*, 1987; USEPA, 1980). At these scales and imaging sensitivity, remotely sensed geodata appeared to be limited for comparisons of AML to biometrics collected at watershed and subwatershed scales (Aronoff, 1993; Richards *et al*, 1994b). Advances in spectral processing may improve this situation. Primarily directly developed geodata based on aerial photography and metadata, this work successfully extended applications of 1:24000 scale geodata and GIS processing to ecological risk assessment frameworks for investigation of coal mining activities.

4.6 CONCLUSIONS

Integrated analysis of habitat-biodiversity, landscape-habitat, and landscape-biodiversity relationships were completed in watersheds with commingled surface and underground mining, urban, and agricultural activities. Covering robust to decimated macroinvertebrate communities, a strong non-linear relationship was observed between taxon richness and abundance biodiversity metrics across multiple land use descriptions in Leading Creek. Subject to departure for elevated abundance in severely stressed subsheds, a power function given by Abundance = Taxa^a with a = 2 for EPT data and a = 1.5 for total data was developed. The relationship encompassed a variety of commingled land uses including agriculture, urbanization, deep long-wall underground mining still active, abandoned strip mining, and abandoned near-surface underground mining.

Results indicated that qualitative benthic macroinvertebrate were reliable indicators of various point source and non-point source anthropogenic stresses in these environs. Further, total, EPT, and respective community strength biodiversity metrics were capable of discerning increasing amounts of stress on subwatershed scales ranging 1 to 1000 km^2, using normalized unit drainage area influences from the landscape developed at a scale of 1:24000. For mining land uses, remotely sensed data was found to be far less reliable in predicting stress to aquatic communities than was directly developed geodata based on aerial photography and metadata. This was true for habitat-landscape relationships analyzed in Chapter 3 and landscape-biodiversity relationships analyzed.

Levels of abandoned underground mining >2% to 5% by area occurring above stream drainage networks lead to significant degradation of aquatic communities. Levels above 15% AUML resulted in decimated ecology due to stream acidification processes with gradation predictable between these levels. EPT taxon richness was decimated in headwater tributaries below the town of Albany with urbanization > 5% by area, where total taxon richness was fairly good. Extremely poor response for EPT taxa in small subsheds observed with urbanization >5% appears to be related to poor habitat and possible water quality concerns. In contrast, total taxon richness was a better indicator of the degree of physical and chemical stress associated with AML land uses than was EPT taxa, though both were responsive. It was shown here that AUML dominates current impacts to biodiversity compared to ASML and active mining. In comparison, active mining influences on habitat and biota were not distinguishable from reference sites. Relations between agriculture and poor biodiversity were also weak. Final coal seam bench status (open = ASML, subterranean = AUML) and concentration of waste piles are two variables still differentiated in ASML and AUML pyretic waste systems. Significant impacts of AUML appear to be attributable to water quality degradation and hydrologic processes associated with AMD moving along bedrock dip above the water table.

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Chapter 5 - COAL BED AND SEDIMENT QUALITY RELATED TO ACTIVE AND ABANDONED COAL MINES IN SOUTHEASTERN OHIO WATERSHEDS

Justin Eric Babendreier

(ABSTRACT)

Examination of acid mine drainage (AMD) related to in situ sediment chemistry was conducted as part of a larger hydrologic-based ecological risk assessment of Leading Creek, located in southeast Ohio. Minimum water column pHs ranged down to 3.3 in smaller 1st order subsheds and upwards of 3.7 to 5.0 in the most heavily AMD impacted 4th order tributary (scale 1:24000). Abandoned mining in Leading Creek involved both underground near-surface mining (AUML) and strip mines (ASML). Active deep long wall mining operations are still conducted in the watershed today and discharge treated pH-neutralized AMD. The analysis incorporated dominant urban, agricultural, and mining land uses normalized to drainage area. Land use/cover threshold criteria were evaluated for AUML=2%, AUML=10%, ASML=3%, Urban=5%, and Bare Soil=3%, the latter representing agricultural activities. Impacts upon aquatic ecology attributed to AUML and ASML from untreated AMD were unrelated to sediment quality. Active mining was associated with the highest concentrations of metals in sediment but had no significant impact on the mainstem of Leading Creek. All sediment concentrations fell within normal ranges of crustal abundance for %C, Fe, Mn, Cu, and Zn.

Ratios of Zn/Fe and Mn/Fe in stream sediments and coal beds also were evaluated for their utility as discriminators of AMD instream. For instream sediments at sites with ASML<3%, LogZn/LogFe ratios were found to be relatively constant across all land uses in Leading Creek as was LogMn/LogFe. High LogMn/LogFe ratios and slightly elevated LogZn/LogFe ratios in sediments were tied to ASML>3%. Unusually low LogMn/LogFe was tied to AUML>10% which exhibited severe AMD impacts and average pH below 6 s.u.. Departures in constant slope relationships for LogZn/LogFe and LogZn/LogFe were not consistently observed for all sites with ASML>3%. Comparatively low with respect to sites with detectable AMD impacts associated with AUML>2%, Mn and Zn were slightly higher in sediment and water columns for the ASML>3% land use category. Non-ASML data sometimes indicated similarly elevated Mn/Fe ratios, implicating sandy acidic soils may determine higher Mn and not actual land uses. Trends between Fe, Zn, and Mn in sediment were not consistently observed in historical sediment studies of southeastern Ohio watersheds. Land use/cover, geologic setting, soil surveys, and chemical quality of mined coal beds were helpful in understanding origin and fate of Fe, Mn, and Zn in stream sediments.

Chapter 5 - COAL BED AND SEDIMENT QUALITY RELATED TO ACTIVE AND ABANDONED COAL MINES IN SOUTHEASTERN OHIO WATERSHEDS

5.1 INTRODUCTION

In Appalachia most acid mine drainage (AMD) is attributed to high sulfur-bearing bituminous coal where oxidation of pyretic sulfides in the coal seam and surrounding rock produces sulfuric acid. According to Skousen (1996), much of the pyrite (FeS_2) in coal and overburden occurs as small crystalline grains intimately mixed with the organic constituents of coal. With lower-sulfur in lignite, subbituminous, and bituminous rank coals in western states, concerns are instead dominated by hard rock mining of copper, gold, uranium and other sulfide ores. Comparing U.S. States, Montana and Alabama have the highest average pyretic sulfur content (0.6%) for lignite rank coals and Utah the highest average content for sub-bituminous coals. For soft bituminous coals, Kansas (3.8%), Missouri and Iowa (2.7%), and Ohio (2.1%) rank highest. Eastward, sulfur content tends to decrease among bituminous coal deposits. Based on the national coal quality database (USGS, 1998), Iowa has the highest pyretic sulfur contents, up to 16%, and levels in eastern Ohio's coal belt have been observed as high as 9%. Pyretic sulfur is the major species and the sulfide of greatest concern in coal beds and associated rocks, and is correlated to the maximum potential acidity of these materials (Skousen, 1996).

In this work, relationships between coal bed quality, surrounding rock and soil materials, and instream sediment quality were investigated. Defining risk on watershed scales requires increased knowledge of major processes and environmental mechanisms associated with active mining, abandoned mined lands (AML), and their respective pyretic acid mine drainages (AMD). The source and final disposition of AMD constituents in sediment is of great interest as it represents both ends of this dynamic continuum. To further understand AMD's impact, instream processes that follow upland pyrite oxidation and acid generation need to be better assimilated into ecological risk assessment frameworks for complex ecosystems like watersheds.

5.1.1 Components of Acid Mine Drainage in Sediments

A classical description of biologically mediated chemical reactions related to pyrite oxidation is summarized well by Nordstrom (1982) and Evangelou (1995). Oxidation of pyrite mobilizes reduced sulfide metals and generates acid, causing dissolution of bedrock and soil materials in water flow pathways. Precipitation of oxides, hydroxides, and oxyhydroxides will typically follow oxidation of metal sulfides, involving a second stage of acid generation. This process can reverse and repeat itself after precipitates reach the instream bed environment through de-acidification processes dominated by sulfate reducing bacteria (Mills *et al*, 1989). Elevated mass concentrations will usually be observed for iron, manganese, and aluminum, and several heavy metals including zinc, nickel, and arsenic among others (Carrucio *et al*, 1974; Evangelou, 1995; Skousen, 1996). Detrimental impacts upon aquatic organisms from AMD have been well documented (Allan, 1995; Newman *et al*, 1991; USEPA, 1980). Excluding physical impacts from inundating sediments, increased acidity with loading and partitioning of

Fe, Mn, Al, and heavy metals instream are believed to impart the major consequences of AMD (Barret *et al*, 1993; Cairns *et al*, 1977; Evangelou, 1995; Nordstrom 1982; Skousen, 1996).

Summarized by Diz (1997) more than a dozen compounds fit within the designations of iron oxides, hydroxides, and oxyhydroxides. Some of the most common are ferrihydrite, goethite, lepidocrocite, hematite, and amorphous ferric hydroxide. Additional minerals form when sulfate is present in solution which usually accompanies AMD instream (Mills *et al*, 1989, Evangelou, 1995). These may include jarosites, melanterite and schwertmannite (Diz, 1997; Nordstrom, 1982, Bigham *et al*., 1994). These minerals differ in important characteristics, including degree of crystallinity, solubility, and thermodynamic stability where amorphous ferric hydroxide appears to be the most common initial solid formed when Fe(III) precipitates (Diz, 1997). With time and loss of water, amorphous solids tend to convert to goethite and hematite which appear to dominate final crystalline forms instream (Diz, 1997; Schwertmann *et al*, 1991).

Major mechanisms of AMD generated from mining activities involve chemical oxidation, microbial oxidation, hydrolysis, neutralization, and precipitation of metal salt solutions. Adsorption of metals to clay and organic carbon and other metal substitutions can occur where Fe, Al, and Mn oxyhydroxides tend to coat sediment surfaces instream (Bigham *et al*, 1996; Diz, 1997; Gray, 1996; Nordstrom, 1982; Robbins *et al*, 1997). Reactions are controlled by several parameters including pH, pE, CEC, solution activities, and common ion effects (Snoeyink *et al*; 1980; Williamson *et al*, 1994). Precipitating oxides are known to be strong scavengers which can remove both cations and anions from solution (Diz, 1997; Benefield *et al*, 1990). Decreases in pH can release cations from mineral surfaces, thus rock, soil, and sediments serve as both sources and sinks of AMD metals found in solution (Diz, 1997; Stumm, 1992).

5.1.2 Coal Bed and Sediment Quality Study Objectives

Attempts to better relate knowledge of AMD chemistry in natural, complex settings were directly investigated here for potential future applications in development of risk assessment frameworks. This was attempted through identification of key *in situ* trends in mineral and heavy metal partitioning in coal beds, stream sediment, and the water column. The hypotheses tested here addressed aspects of hydrologic-based watershed-scale ecological risk assessment by employing GIS based land use/cover analysis. The underlying objective was to first evaluate quantitative descriptions of urban, agriculture, and surface and underground coal mining, relating land use/cover to instream sediment chemistry. Both active and abandoned mining activities were considered. Few studies of active and abandoned mining exist that combine Geographical Information System (GIS) at fine scales of 1:24000, land use/cover analysis, and risk assessment. Work reported here was developed as initial tasks in testing the following two hypotheses: (1) Impact upon aquatic ecology attributed to AMD is a function of water quality degradation and not sediment quality and (2) The ratio of zinc to iron in sediment is relatively constant across urban, agriculture and coal mining land uses, but the ratio of manganese to iron is not.

This paper initializes a discussion on sediment quality processes and their relationship to pollutant loading from the landscape. Due to their complexity, critical aspects supporting the stated hypotheses were investigated in this paper by first comparing land use/cover variables to measures of sediment chemistry, focusing on how sediment quality is related to the landscape and coal bed quality for AMD metals. The first hypothesis extends hypotheses tested in Chapters 3 and 4. There it was shown that severe AMD tied to near-surface abandoned underground mine lands (AUML) decimated aquatic biodiversity. Similarly, AMD was not attributed to increased degradation of habitat associated with abandoned strip mines (ASML). The first hypothesis stated was also borne out of early work in various watersheds including Leading Creek that indicated a surprising lack of sediment toxicity at severely impacted AMD sites (Babendreier *et al*, 1997). Discussed in Chapters 7 and 8, a three-tiered ecotoxicological testing strategy spanning sediment and water columns completes evaluation of the first hypothesis.

In testing the second hypothesis stated, low Mn/Fe is also shown to be associated with severe AMD. Unusually high Mn/Fe is attributed to disturbance of soils usually associated with ASML. The second hypothesis is further evaluated in Chapter 6 where iron and zinc sediment ratios are shown to be associated with co-precipitation of AMD pollutants and water column suspensions of fine silt/clay size materials. In Chapter 6, manganese solid-liquid phase dynamics are also further described with respect to coarser sandstone sediment particles where paucity of Mn in sediments at sites with severe AMD is attributed to instream dissolution.

5.2 STUDY WATERSHEDS

Leading Creek is a fifth order stream (scale: 1:24000) 96% of which spans Meigs County, with other portions in Athens, and Gallia Counties. Together, 11 mainstem stations and 19 tributary stations shown in Figure 5.1 in were selected for hydrologic, hydraulic, and ecological risk assessment monitoring. Mainstem stations were located to provide complete longitudinal coverage of Leading Creek, and discrimination among potential sources of sediment or contaminant impacts to the stream system. Monitoring included 3 reference sites

5.2.1 Leading Creek Mainstem and Tributary Stations

Ten mainstem sites and six of the tributary sites were included in stream stage monitoring indicated by designation as staff gage stations in Figure 5.1. Agricultural activities dominate the upper mainstem while downstream reaches are dominated by mining. Eighteen tributary stations in the Leading Creek Watershed received some level of consistent sampling effort for most variables described here. These stations were initially chosen based on association with either (1) their known or suspected role in the contribution of point source pollutant loading to Leading Creek, or (2) their significance based on the size and nature of associated drainage area. The six tributaries selected for stream stage monitoring were chosen based on proximity to significant points on the mainstem of Leading Creek, proximity to known point sources and suspected nonpoint sources, or based on their significant representation of major tributary inflows. Parker Run (TS11) and Ogden Run (TS6) in Figure 5.1 receive active coal mining treatment plant effluents and assorted stormwater and sanitary discharges. Other NPDES sites relate to sanitary treatment (STP), water treatment (WTP), and quarry discharges.

5.2.2 Ecoregion Reference Watersheds

Three comparative reference stream sites in two watersheds were selected on the basis of (1) exceptionally high IBI/ICI biodiversity scores in historic assessments, or (2) visibly exceptional habitat characteristics that are representative of what might be attainable in the Western Allegheny Plateau ecoregion. IBI refers to the site's Index of Biotic Integrity, a fish-based metric, and ICI refers to the Invertebrate Community Index, an insect-based metric (Karr, 1991; Miller *et al*, 1988; OEPA, 1989; USEPA, 1996). Upon initial inspection, the reference sites did not indicate marked decline but were potentially impacted by various anthropogenic traits including adverse effects from urban, agricultural, ASML, or AUML sources. Due to the wide reach of AML in Appalachia, there were no readily identifiable similar sized watersheds nearby that could claim strong independence from all or most of these stresses.

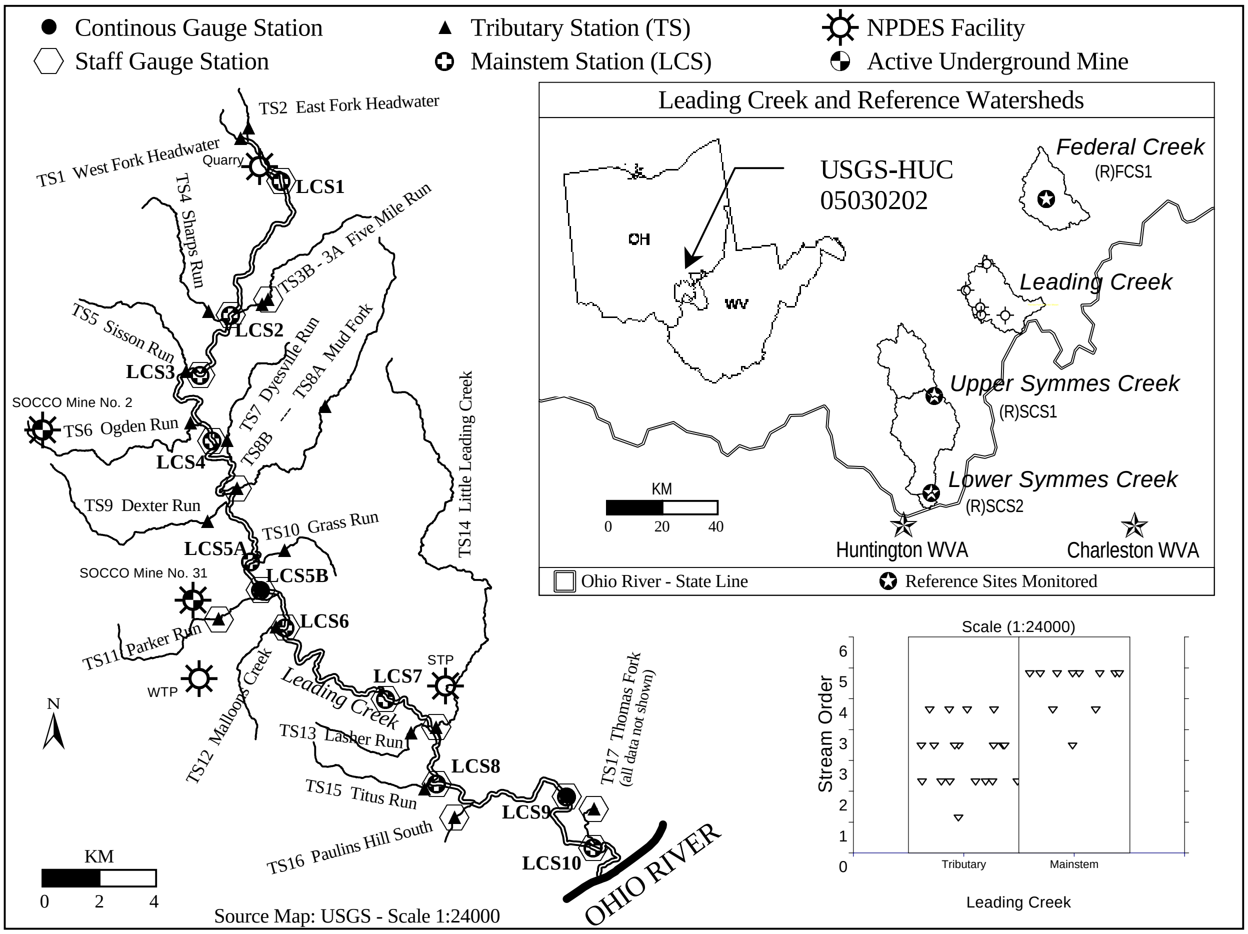

Figure 5.1 - Leading Creek Monitoring Network and Reference Watersheds

Federal Creek, a primary tributary of the Hocking River, spans Athens and Morgan Counties and drains 375 km^2 with mainstem length of 38 km. The subwatershed area monitored was 68 km^2 comprising the northwestern headwaters above Amesville, Ohio on State Route 329 at County Road 35. The average energy line fall of Federal Creek mainstem is 0.36%. Federal Creek has mostly ASML influences based on review of U.S. Geological Survey (USGS) land use coverages (scale 1:24000), where upper reaches were noted by OEPA to have minimal AMD impact. Symmes and Leading Creeks are both within Ohio EPA's study group of southeast tributaries draining to the Ohio River. Symmes Creek covers portions of Lawrence, Jackson and Gallia Counties, and is 113 km in length with slope 0.064% and drains an area of 920 km^2. The upper and lower reference sites each drain 348 km^2 and 897 km^2, respectively. The lower site is near State Route 243 below McKinney Creek, and the upper site near State Route 141, above Sand Fork. Upper Symmes Creek is more influenced by ASML and AUML than the lower site.

5.3 METHODS AND MATERIALS

Data collection in Leading Creek and the reference watersheds included monitoring of: (1) Land use and land cover, (2) precipitation, stream stage, water velocity, and flow; (3) sediment and water quality; (4) benthic macroinvertebrate and stream habitat; and (5) toxicity through sediment, acute, and *in situ* testing programs. To assess chemical, physical, and biological interactions, stations in Figure 5.1 were examined over a wide range of drainage areas and stream orders (Minshall *et al*, 1985; Strahler, 1957; Vannote *et al*, 1980; USEPA, 1996). Data collection methods specific to measurement of water quality indicators, sediment quality, and land use/cover are detailed or outlined here. Coal bed quality data was derived from the USGS CoalQual Database Version 2.0 (USGS, 1998).

5.3.1 Land Use and Land Cover

Spatial analysis of subsheds used several geodata sources including data hand-digitized from USGS quadrangle 7.5-minute map series and USGS hydrology geodata obtained from Ohio Division of Natural Resources (ODNR). Land use/cover data (scale 1:24000) was developed through three primary sources including a) 7.5-minute quadrangle series maps depicting abandoned strip mines, b) Ohio Division of Natural Resource (ODNR) geodata depicting abandoned underground mines, and c) 1988 Landsat5 Thematic Mapper (TM) geodata developed by ODNR specifically for assessment of abandoned mine lands. Described in Table 3.1 in Chapter 3 and in Appendix A, definitions, processing and interpretation of 1988 TM data were provided by ODNR. Drainage areas for monitored tributaries span 124 to 8038 ha.

5.3.2 Water Quality Indicators

Water quality indicators were analyzed in this analysis to impart quantitative delineation between various point source and non-point source influences in Leading Creek. Water quality sampling was conducted at all mainstem, tributary, and reference sites. Monthly water column grab samples for pH, specific conductivity, and water temperature were collected and analyzed in the field. Grab sampling at each stream site was conducted using a clean plastic bucket tethered to a rope and dropped from bridge sites into the swifter areas of flow. The bucket was rinsed with site water prior to use each time. Water column samples used standard water and wastewater methods (APHA, 1992). Specific conductivity was normalized to 25 ^oC (1.91% standard temperature coefficient; $\pm$ 1 umho/cm) and water temperature ($\pm$ 0.01 ^oC). Portable pH probes were calibrated with pH 4, 7, and 10 lab-grade buffers and conductivity was calibrated with standardized KCl solutions using lab-grade deionized water (APHA, 1992).

5.3.3 Sediment Chemistry

Sediment chemistry sampling was conducted at all mainstem and tributary sites, and the Federal Creek reference site. Annual fine sediment samples were collected in 1996 and 1997 using standard grab sampling techniques in accordance with standard methods for toxicity testing (ASTM, 1990; APHA, 1992). At each site 3 to 5 replicate samples were collected from a representative depositional area along a cross-sectional transect, restricting sampling to the upper 1 to 10 cm of bed depth (Burton *et al*, 1992). Water and sediment depths were collected for sediment samples during each sampling event (Gallagher *et al*, 1999). Sediments were collected by wading cross-sections except in 1996 stations LCS9 and LCS10 required use of an Ekman dredge lowered from the bridge due to higher water depths. Samples were placed in sterile bags and returned on ice to the Environmental Laboratory Systems facility at Virginia Tech where they were refrigerated at <4 ^{o}C until analysis was completed.

Due to two-week holding times for associated sediment toxicity tests, the watershed was split into upper, middle, and lower sections and sampled on three different dates each year. Historical data was also used to supplement annual sediment chemistry data for the reference watersheds and at available sites within Leading Creek (OEPA, 1991a, 1991b; Birge *et al*, 1995). Sediment samples in 1996 and in most cases for 1997 were split for sediment toxicity testing and chemical analysis was completed for %carbon, Fe, Mn, Zn, and Cu. For some sites, 1997 sediment toxicity samples were collected separately from a second round of samples collected in early 1998 for chemical analysis. These are referred to here as 1997 samples. Testing of total metals in solid samples collected were analyzed using acid extraction followed by analysis with inductively coupled plasma (ICP) via USEPA Method 220.7. Sediment samples were kept cool (<4 ^{o}C) and transported to American Electric Power's Dalton laboratory in Columbus, Ohio for analysis. The Dalton lab facility is used by AEP to meet NPDES permit compliance monitoring requirements for their regional fuel extraction and power generation facilities.

5.3.4 Description of Statistical Tests

All parametric normality tests described here included skewness, kurtosis, and combined omnibus tests (D'Agostino *et al*, 1990; Zar, 1999) available within the NCSS97 software package (Hintze, 1997). Where used, parametric ANOVA between group means was supplemented using the Tukey-Kramer multiple comparison test with α_{MC} = 0.10. In cases of non-normality of residuals or heterogeneity of variance, Kruskal-Wallis one-way ANOVA ranks were performed testing hypotheses stated on group medians and used data corrected for ties. For non-parametric ANOVA, the Kruskal-Wallis multiple comparison z-value test was used to evaluate significant ANOVA results employing the full Bonferroni criteria for all groups (Hintze, 1997; Zar, 1999). Power stated is for associated parametric test results where Zar indicated that power of the non-parametric procedure is normally $\pi/3$ or 95% of parametric results. Unless stated otherwise, a significance level α = 0.05 was used in all statistical analyses.

5.4 RESULTS

5.4.1 Land Use and Land Cover Analysis

Land use/cover variables addressing urban, agricultural and mining activities are summarized in Table 5.1 and can be viewed graphically in Figures 3.2 through 3.7 in Chapter 3. Characteristics and qualifications concerning geodata used here are described in more detail in Chapter 3 and Appendix A. Leading Creek coverage of abandoned underground mines in Table

5.1 represents 124 non-overlapping coal mine works where all reside above stream drainage elevations. In Figure 3.7, long wall mining operation aereal extents shown (>100m BLS) at Mines Nos. 2 and 31 cover 6300 ha and represent deep groundwater diversion zones associated with treated AMD discharged into Leading Creek. These areas were not included in AUML totals in Table 5.1. Excluding historic and current SOCCO operations, most underground mining in the Leading Creek vicinity was conducted prior to 1910, primarily during the latter portions of the 19th century. Most strip mining occurred after WWII where 80% or 1379 ha of strip mines in Leading Creek were created by 1961, and 20% or 353 ha between 1961 and 1978. Little surface mining occurred after this period due to the Surface Mining and Reclamation Act of 1977.

Table 5.1 – Summary of Directly Sensed and Remotely Sensed Land Use and Land Cover

Station ID	Directly Digitized			June 1988 TM	
	Total ASML	Recl. ASML	Total AUML	Urban	Bare Soil
(R)FCS1	2.1%	0%	0.06%	-	-
(R)SCS1	3.8%	0.5%	0.97%	-	-
(R)SCS2	2.6%	0.3%	0.44%	-	-
LCS1	0%	-	0%	4.0%	8.3%
LCS2	0%	-	0%	1.2%	4.4%
LCS3	0%	-	0%	0.9%	4.4%
LCS4	0%	-	0%	0.5%	4.6%
LCS5B	1.0%	0.4%	0%	0.3%	2.9%
LCS6	0.9%	0.3%	0%	0.2%	2.8%
LCS7	1.1%	0.3%	0%	0.2%	2.5%
LCS8	3.3%	1.0%	0.53%	0.1%	2.1%
LCS9	3.8%	1.0%	0.93%	0.2%	2.1%
LCS10	4.5%	0.8%	4.16%	0.1%	1.9%
TS1	0%	-	0%	12.7%	7.9%
TS2	0%	-	0%	7.0%	6.7%
TS3B	0%	-	0%	0%	3.8%
TS4	0%	-	0%	0%	5.8%
TS5	0%	-	0%	0%	2.9%
TS6	0%	-	0%	0%	6.4%
TS7	0%	-	0%	0%	0.6%
TS8A	5.5%	1.5%	0%	0%	2.3%
TS8B	3.6%	1.4%	0%	0%	1.5%
TS9	0%	-	0%	0%	1.1%
TS10	5.3%	1.9%	0%	0%	0.1%
TS11	0%	-	0%	0%	2.3%
TS12	0%	-	0%	0%	0.2%
TS13	11%	0%	0%	0%	0.6%
TS14	8.5%	3.7%	1.67%	0%	1.0%
TS15	15%	0%	2.79%	0%	1.3%
TS16	30%	0%	19.5%	1.3%	0.2%
TS17	6.6%	0.3%	15.9%	0%	1.0%

5.4.1.1 Geologic Setting of Leading Creek

The geologic setting for the study area is within the Interior Lowland topographic province. Geology in Leading Creek is described by horizontally oriented sedimentary bedrock. Beds are characterized by northeast-southwest strike and average dip of 5.7 m/km towards the southeast (Lucht *et al*, 1985). Underlying geologic units include members of the Pennsylvanian age Conemaugh Group (Pc) and Allegheny Formation (Pa). Depositional environment varies between marine and non-marine conditions, and is distinguished by multiple coal sequences. The surface lithologies vary between the Pennsylvanian age Monongahela Group (Pm) and the Pennsylvanian/Permian age Dunkard Group (Pd). The depositional environment was a non-marine Pennsylvanian – Permian basin also distinguished by multiple coal sequences, with likely minor amounts of limestone (Cardwell, 1968; Owens, 1967; Summerson, 1962).

Bedrock associated with surface and underground mining conducted above stream channel elevations in Leading Creek lies within the Monongahela Group. The Monongahela Group is an estimated 90 m thick sequence of non-marine sandstones, siltstones, shales, and coal subdivided in some locations into the Uniontown and Pittsburgh Formations, including the near surface Redstone coals mined in Leading Creek. The upper lithologies in Leading Creek's ASML mining district drop from massive sandstone cap rock through a 1 m shale layer lying above the Redstone #8A coal bed. The geology would be expected to offer low to medium carbonate mineral buffering capacity to offset pyrite oxidation (USEPA, 1980; Skousen, 1996). Spatial variation in near-surface bedrock of Leading Creek subsheds with ASML is expected to be minimal, except for thinning towards the northwest. Next to bedrock's overall dominance, soil and vegetative type can exert significant secondary influence on stream sediment and water column acidification processes (USEPA, 1980; Richards *et al*, 1996; Daniels *et al*, 1995)

Excluding unreclaimed strip mined lands, dominant soil series encountered across Leading Creek's hillside and channel-valley morphologies (Schumm, 1977) are silt loam texture classes (Gillmore *et al*, 1991) with minor exceptions. The silt loams encountered have a typical silt particle size content in the range of 60-85%. The disturbed, unreclaimed ASML lands are typified by sandy and sandy loam soils, comprised of a mixture of original silt loam series, capstone, and portions of underlying bedrock. Associated disturbed ASML sands and sandy loams range 65-100% sands (diameter <2 mm), with a median diameter D_{50} in the medium to fine particle size range (0.125-0.5 mm). Digital soils data for Leading Creek were not available. Summarized by Evangelou (1995), in the eastern United States coal fields mine waste clay mineralogy is expected to be dominated by kaolinite, mica, quartz, and chlorite.

Comparison of soil series was made among disturbed strip mines (Pn), reclaimed strip mines (Pu), and predominant undisturbed hillside surface soils in the mining district (Ug). The unreclaimed Pinegrove (Pn) is associated with USDA's hydrologic soil group with high infiltration, and the Upshur-Gilpin (Ug) complex and reclaimed Pinegrove series (Pu) are both characterized by slow to very slow infiltration rates. Disturbed ASML soils have higher percentages of gravel range particles (>2 mm) compared to background silt loams. In areas with ASML activity defined by unreclaimed Pinegrove (Pn) soil series in the Upshur-Gilpin-Pinegrove association, soil reactivity is the highest with extreme to strong soil reaction pH of 3.6 to 5.5, which also characterizes the undisturbed Gilpin soil series. Slightly higher pH of 4.5 to 6.5 is attributed to the Upshur and reclaimed Pinegrove series. Other soils of Leading Creek are characterized by higher soil pH of 4.5 to 8.4 (Feusener *et al*, 1989; Lucht *et al*, 1985). These data point out unique differences in the ASML mining belt soil and bedrock characteristics.

5.4.2 Water Quality Indicators

In comparison to active underground mining, minor point source permitted National Pollutant Discharge Elimination (NPDES) discharges given by the WTP, STP in Figure 5.1 and sanitary wastewaters from the active mines represent insignificant loading of treated effluents and flow into the stream system. The active mining operations must meet categorical effluent limitations established for coal treatment plants (USEPA, 1976; 40 CFR 434). Treatment of coal preparation wastewater involves primarily physical separation processes and pH neutralization using sodium hydroxide. Waste sludge results from precipitates of primarily iron, manganese, and aluminum oxyhydroxides and co-precipitation of heavy metals (Diz, 1997; USEPA, 1976). Coal preparation plant wastewater effluent is high in total dissolved solids (TDS) primarily composed of sulfate, sodium, and chloride where conductivity at SOCCO Mine No.31 typically ranges between 4.0 to 6.2 mmhos/cm.

During lower flow conditions the watershed can gain significant alkalinity from deep long wall mining mine operations, helping buffer AML-AMD entering the lower mainstem. The quarry discharge is also a significant contributor of headwater alkalinity during low and medium flows. For most periods of flow, Mine No.2 wastewaters are conveyed to Mine No. 31 for treatment and storage before final discharge to Parker Run (TS11). Following periods of increased groundwater infiltration after run-off, some process wastewaters are treated and discharged to Ogden Run (TS6). Figures 5.2a and 5.2b present summarized data for water column pH and conductivity across Leading Creek, including combined data for reference watersheds Federal Creek and Symmes Creek. This data was taken from monthly water quality monitoring results conducted between April 1996 and October 1997. Sites were combined based on land use/cover geodata to assist in delineating significant differences.

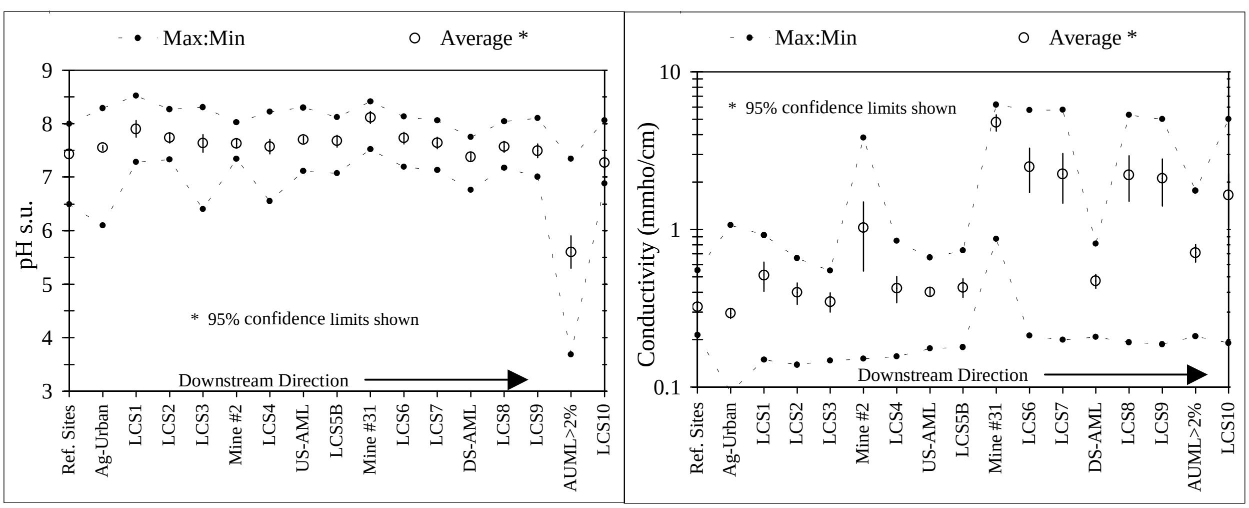

Figure 5.2a and 5.2b – Average (a) pH and (b) Specific Conductivity - 1996 to 1997

Qualitative land use/cover designations in Figures 5.2a and 5.2b are defined by a) Ref Sites: (R)FCS1, (R)SCS1, (R)SCS2; and for Leading Creek tributaries b) AG-Urban: TS1 to TS5, TS7, TS9, and TS12; c) Upstream (US)-AML: TS8A, TS8B, TS10; d) Downstream (DS)-AML: TS13 and TS14; and e) AUML>2%: TS15, TS16, and TS17. Station TS15 lies above LCS8 and TS16 above station LCS9, but AMD loading from these tributaries are not typically discernable in mainstem water quality data. With average pH 6.7 and 7.3, respectively, TS15 and LCS10 were less impacted by AMD and can be described by 2%<AUML<10%. TS16 and TS17 had average pH 4.6 and 5.5, respectively, with AUML>10%. LCS10 is the only mainstem site with AUML>2%. Sites with mild and severe AMD were all distinguished by AUML>2%.

The limestone quarry above LCS1 is responsible for increased pH and conductivity observed at that station. Seen in Figure 5.2a, significant differences in pH were found for sites impacted by AMD, TS11, LCS1, and other stations. Specific conductivity also highlighted differences between untreated AMD impacted reaches with moderate levels, Mine Nos. 2 and 31 and lower mainstem sites with the highest levels, and decreasing levels from LCS1 in the upper mainstem attributed to discharges from the closed quarry. A remarkable distinction in Leading Creek, ASML was unrelated to the degree of stream acidification compared to AUML.

5.4.3 Sediment and Coal Bed Metal Chemistry

Two sampling events per site were conducted and sediment chemistry was evaluated for %carbon, Fe, Mn, Zn, and Cu. Cross-section distributions of sediment chemistry by year are presented in Tables H1 and H2, respectively, in Appendix H.

Shallow underground mining and strip mining in Leading Creek were dominated by excavation of the Redstone #8A coal bed also referred to as the Pomeroy #8A in Ohio, and the Redstone #8 in other areas and states. Minor amounts of coal if any were likely extracted from the lower Conemaugh Group. Below this formation, active SOCCO deep mines work the Clarion #4A coal bed in the Allegheny Formation. Statistics developed here based on USGS's CoalQual Database (USGS, 1998) indicated slightly higher pyretic sulfur and total sulfur were found in surface lithologies associated with the Redstone #8A coals compared to the Clarion #4A. Table 5.2 provides background chemistry on the Redstone #8A coal bed associated with ASML and AUML in Leading Creek, and the Clarion #4A bed associated with deep long wall mining at SOCCO's active Mines Nos. 2 and 31. Chemical constituencies were evaluated for comparison to sediment contaminants found instream.

Table 5.2 – Geochemistry of Mined Coal Beds of Leading Creek

Coal Bed	Ohio County	%Ash	% Pyretic Sulfur	Whole Coal Elemental (ppm)		
				Fe	Zn	Mn
Clarion #4A	Meigs	26%	4.3%	60,000	22	39
"	"	17%	1.1%	13,000	23	31
"	"	23%	1.5%	23,000	33	26
"	Vinton	24%	2.0%	30,000	16	75
"	"	15%	1.2%	19,000	17	48
Redstone #8A	Gallia	9%	2.8%	29,000	17	30
"	"	11%	3.5%	28,000	36	16
"	"	14%	3.0%	40,000	22	34
"	"	20%	3.3%	64,000	31	39
"	"	11%	2.2%	28,000	12	18
"	Lawrence	24%	4.9%	61,000	33	87
"	"	8%	2.0%	28,000	12	19

The Redstone #8A samples in Table 5.2 represent the most locally available samples with two samples from Lawrence County and 5 from Gallia County. Other Redstone #8A coal bed samples in West Virginia and Pennsylvania and to the northeast in Ohio had similar levels of Zn and Mn except lower Fe and pyretic sulfur could be observed, consistent with cleaner coals in those regions. Due to a marine depositional origin, higher sulfur and alkalinity might be expected in the Allegheny Formation compared to surface lithologies (Caruccio *et al*, 1974; Younger, 1996), though this was not consistently the case in Leading Creek. Cecil (1985) pointed out that high variability in sulfur and ash content of younger coal beds including those within the Monongahela Formation are a reflection of variable geochemical conditions of non-marine sedimentary environments. Cecil also found a slightly higher average total sulfur content in the Monongahela (3.4%) compared to the Charleston/Allegheny (2.9%). The Redstone #8A exhibited typically higher levels of pyretic sulfur content than the 2.1% average ($\sigma = 1.3$; $n = 660$) for all bituminous coal samples in Ohio's eastern mining belt (USGS, 1998). A higher pyretic sulfur content is indicative of more predictable problems from AMD, especially when the host geology is known to be low in buffering capacity (Skousen, 1996; USEPA, 1980)

All samples in Table 5.2 were listed as channel-type samples in the CoalQual database indicating opened bench or road cuts. Analytical techniques for data in Table 5.2 included whole coal conversion of ash remaining after ignition at 525 ^{o}C, and electronic spectrographic analysis (Mn) or atomic adsorption analysis (Mn, Zn), and x-ray fluorescence analysis (Fe) on ash. Interspersed inorganic materials are identified by the percentage of ash remaining after ignition. Coal bed samples analyzed by USGS by default exclude all data with ash >33% and attempt to reflect the purest portions of coal beds in locales sampled. Skousen (1996) indicated for Appalachia that the character of pyretic contents in materials in and around coal beds reflects to a large degree the pyrite that is oxidized after extraction. With uncertainty not quantifiable, relative levels are expected to reflect content of bedrock in close proximity to opened benches.

Strip mining in southeastern Ohio typically removes coal and overburden down to a clay/shale floor, leaving the resulting bench exposed. With respect to pre-disturbance, in Leading Creek most of the source pyrite in AUML and ASML generating AMD that reaches the stream system is expected to originate from shales at, above or close to the extracted coal bed bench. ASML disturbance in Leading Creek is roughly ½ the age of major AUML disturbances. Final bench status (open = ASML, subterranean = AUML) and concentration of waste piles are two variables still differentiated in ASML and AUML pyretic waste systems in Leading Creek. This last statement excludes those areas in Table 5.1 designating reclaimed strip mine lands.

5.4.4 Sediment Quality Assessment

Inter-annual analysis of sediment chemistry in Leading Creek between 1996 and 1997 is presented in Table 5.3. For reference, comparison of water and sediment quality in Leading Creek to average stream and crustal abundance is provided in Table 5.4. Abundance data for global earth in Table 5.4 was taken from Winter (2000) who developed the average stream water values (Porterfield, 1984; Huheey, *et al*, 1993), and average crustal abundance values (Butler, *et al*, 1989; Greenwood *et al*, 1997; Huheey, *et al*, 1993; James, *et al*, 1992; Kaye, *et al*, 1993; McGraw-Hill, 1992; Porterfield, 1984). Pre-empting discussions in Chapter 6 on water column results, iron was strongly correlated with Total Suspended Solids (TSS) for sites with minimal

AMD impact, explaining high water column iron concentrations in Table 5.4. Tied with severe AMD impact in Leading Creek, elevated zinc and manganese in the water column usually followed low pH. For Leading Creek, water column carbon was represented by Total Organic Carbon (TOC) data. In Table 5.3, inter-annual results indicated greater temporal variation in sediment chemistry for carbon and especially copper in upper watershed tributaries and mainstem reaches with increased agricultural activities. Power (1-β) of the null hypothesis test for zero regression slope was 1.00 for all but Cu which was 0.06.

Table 5.3 – Regression of Inter-Annual Log-Normal Sediment Quality Data – Leading Creek Station Results for 1996 to 1997

Parameter	R^2	[P]
%Carbon	0.51	4.4×10^{-5} *
Copper	0.04	0.75
Iron	0.74	$<1\times10^{-6}$ *
Manganese	0.59	3×10^{-6} *
Zinc	0.71	$<1\times10^{-6}$ *

Shown in Table 5.4, for all sites including TS11, and for all parameters, sediment concentrations observed were not elevated with respect to crustal abundance concentrations. Levels were not consistently distinguishable by land use other than elevated metals noted for active mining and elevated %carbon for smaller urbanized subsheds. Except for station TS11 below Mine 31, the reference site Federal Creek had relatively higher iron sediment concentrations compared to most Leading Creek stations. Next to TS11, stations TS1, TS2, LCS1, TS10, and TS12 had typically the highest levels of Fe, Zn, and Mn inside the watershed. TS1 and TS2 are heavily urbanized where LCS1 drains these two sites and is also the most strongly influenced by the closed limestone quarry. TS10 represents the youngest ASML in the watershed, and TS12 is subject to some agriculture activity with livestock operations.

Table 5.4 – Reference Stream Water and Sediment Abundance Data Compared to Leading Creek Station Results for 1996 to 1997

Chemical	Average Stream Water Abundance (ppb)	Range of Water Quality Data on Leading Creek (ppb)	Average Crustal Abundance (ppm)	Range of Sediment Quality Leading Creek (ppm)	Average (σ) Sediment Quality Leading Creek (ppm)
C	1200	1000 – 10,000	0.18%	0.04% - 2.5%	0.50% (0.48%)
Cu	6	1 – 30	68	1– 93	21 (16)
Fe	670	60 – 38,000	63,000	5,800 – 94,000	24,000 (16,000)
Mn	5	20 – 17,000	1,100	150 – 2,400	750 (510)
Zn	10	4 – 750	79	13 – 150	53 (28)

5.4.4.1 Analysis of Land Use/Cover

Using pooled annual data, non-parametric ANOVA testing (n=55 to 57; α = 0.05) indicated TS11 and Federal Creek had significantly higher Cu ([P = 0.014]; 1-β = 0.71) than sites below Mine 31, and higher Fe ([P = 0.0042]; 1-β = 1.00) than sites above (i.e. not influenced by) and below Mine 31. No differences for Fe and Cu though could be discerned between TS11 and the reference site. Using non-parametric tests again, at TS11 Mn ([P = 0.026]; 1-β = 0.78) was higher than sites above and below Mine 31, and Zn ([P = 0.017]; 1-β = 0.87) was higher than sites above Mine 31. In the latter case, parametric testing concluded higher levels also between TS11 and sites below Mine 31 ([P=0.0059]), where an assumption of unequal variance was rejected (α = 0.05; [P=0.025]). These tests indicated only a localized influence of active Mine No. 31 on sediment quality, where no mainstem impact was discerned.

Differences between TS11, sites above and below Mine 31, and Federal Creek were not observed in parametric testing on %carbon which meet assumptions for equal variance and normality ([P = 0.43]; 1-β = 0.73). No groups were significantly different than the reference site Federal Creek for parameters Zn and Mn. For evaluation of threshold land use criteria Urban=5% and Bare Soil=3%, differences were noted again for Fe between Leading Creek stations and Federal Creek. Other points of interest in parametric ANOVA tests were that Bare Soil>3% had lower Zn than Bare Soil<3%, Urban>5%, and the reference site ([P = 0.0092]; 1-β = 0.83), and Urban>5% had higher %carbon ([P = 0.026]; 1-β = 0.73) than both Bare Soil categories, but not the reference site. Analysis of mining land uses with criteria ASML=3% and Urban=5% found similar patterns for carbon and indicated that Zinc was lower in subsheds with ASML<3%, which encompassed agricultural sites with Bare Soil above and below 3%. Carbon content >0.6% to 0.8% was preferentially associated with subsheds <20 km^2, but this threshold was not a significant indicator of elevated %carbon (n=28; α = 0.05; [P = 0.46]; 1-β = 0.36).

In Table 5.5, describing linear regressions among sediment parameters tested, zinc, manganese and iron were the most closely correlated. Data is presented for sites partitioned by a criteria of ASML=3% by surface area. As shown, copper in sediment was weakly related to Fe, Mn, and Zn. Log transformed annual copper data was also found to be unrelated to %Bare Soil (n=54; R^2 = 0.002 [P = 0.75]; α = 0.05, 1-β = 0.061) and %AG/Urban land use/cover data (n=54; R^2 < 0.001 [P = 0.93]; α = 0.05, 1-β = 0.05), the latter regression failing residual tests for normality. In other unreported work, it was found in water column load analysis that Bare Soil and AG/Urban land uses were related to elevated TSS for high flows, but not to copper variation. In Table 5.5, power (1-β) of the null hypothesis test for slope=0 in the regression was between 0.72 and 1.00 for significant test results and between 0.10 and 0.40 for insignificant tests.

Table 5.5 – Linear Regression of Log-Normal Sediment Quality Data – Leading Creek Station Results for 1996 to 1997

Chemical	%Carbon		Copper		Iron		Manganese	
	R^2	[P]	R^2	[P]	R^2	[P]	R^2	[P]
Cu; ASML>3%	0.006	0.72	-	-	-	-	-	-
ASML<3%	0.21	0.0079 *	-	-	-	-	-	-
Fe; ASML>3%	0.26	0.014 *	0.24	0.015 *	-	-	-	-
ASML<3%	0.45	$1.9x10^{-5}$ *	0.27	0.0019 *	-	-	-	-
Mn; ASML>3%	0.03	0.45	0.021	0.50	0.037	0.37	-	-
ASML<3%	0.48	$7x10^{-6}$ *	0.32	$6.7x10^{-4}$ *	0.87	$<1x10^{-6}$ *	-	-
Zn; ASML>3%	0.31	$7.3x10^{-3}$ *	0.12	0.09	0.39	0.0010 *	0.077	0.19
ASML<3%	0.54	$<1x10^{-6}$ *	0.33	$4.7x10^{-4}$ *	0.94	$<1x10^{-6}$	0.83	$<1x10^{-6}$ *

5.4.4.2 Coal Quality and Mining Land Uses

Statistics in Table 5.5 present linear regression and associated ANOVA results. Of all sediment chemistry combinations evaluated, Zn and Fe, Mn and Fe, and Mn and Zn were the most strongly correlated for sites with ASML<3%. Similar but weaker results were found if AUML was partitioned, and poorer results were also attained by considering only unreclaimed ASML. For many sites with ASML>3%, Mn, and Zn to a lessor degree tended to rise relative to Fe. Figures 5.3a and 5.3b summarize these findings for ASML. For all data in Tables 5.3 and 5.5 and Figures 5.3a and 5.3b, log–normal transformations were used to attain normality of residuals and $\alpha = 0.05$ was used to test significance. Not distinguishing AUML>10%, curves in Figure 5.3 lumped TS16 and TS17 into ASML>3%. Average annual data was combined and comparisons of subsection point data were omitted to avoid strong serial correlation exhibited. Residuals for %C and Cu, Fe and Cu, and Fe and Zn for %ASML>3% were not normally or log-normally distributed, where %C and especially Cu exhibited weak inter-annual correlation.

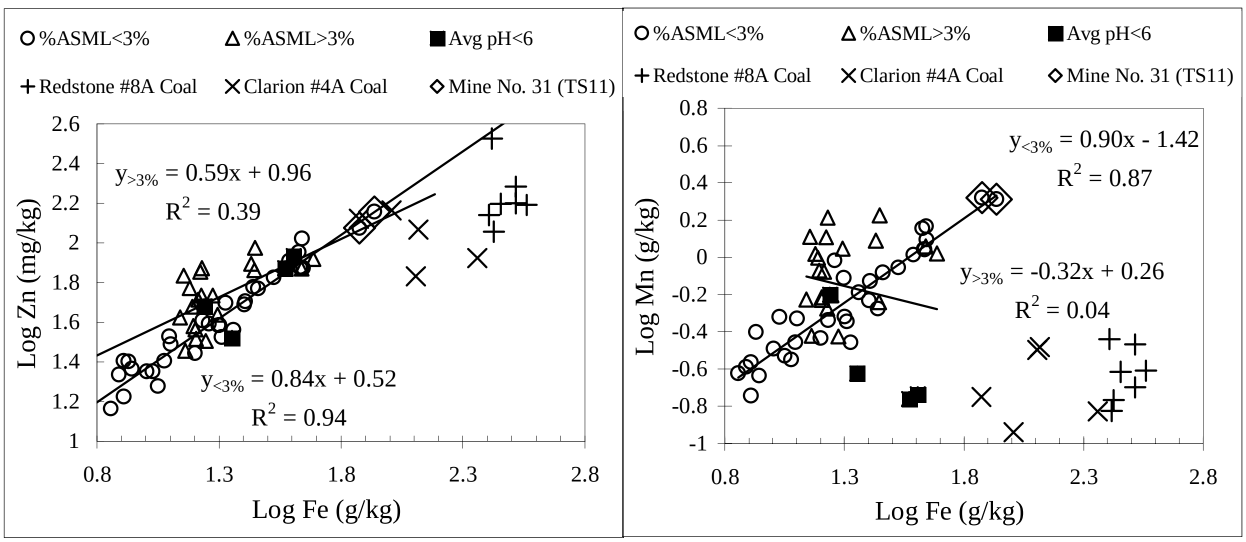

Figure 5.3a and 5.3b – (a) Zinc and (b) Manganese versus Iron Chemistry in Sediment – Leading Creek Station Results for 1996 to 1997

Shown more clearly in Figure 5.3a and 5.3b, TS11 with strong active Mine No. 31 influence had the highest Zn, Mn, and Fe values compared to all sites in Leading Creek. In Figure 5.3a, Zn/Fe ratios in the sediment column did not discriminate among land uses, with a strong correlation observed between Fe and Zn. The two samples for Clarion #4A identified most closely with Zn and Fe content in TS11 were collected from the Meigs Mine No.2 area.

Analysis in Figure 5.3b also identified a relative paucity of manganese in sediment at the most severely impacted AMD sites with AUML>10%. At the two severely acid stressed sites in Leading Creek, TS16 and TS17, lower Mn/Fe ratios did not directly trend with lower average pH. Underscored for Mn and Fe relations in Figure 5.3b, coal beds also showed a paucity of Mn, indicating absence of sandstones (Robbins *et al*, 1997). In the case of Mn/Fe ratios in sediment, however, described by TS11, Mine No. 31 extracting the Clarion #4A had correlated levels of Mn and Fe, similar to other sites with ASML<3%. Mine No. 31 discharges treated AMD removing most Fe, Zn, and Mn by increasing pH and precipitating Fe, Mn, Zn, and other metals (USEPA, 1976). Seen in Figure 5.3a coal bed Fe and Zn content was elevated with respect to all sediments. Levels of Zn in ash were still within average range of crustal abundance data shown in Table 5.4. Fe was only modestly elevated for some samples in Table 5.2, with slightly above average abundance levels for the dirtier Redstone #8A with higher pyretic sulfur contents.

An evaluation of log transformed sediment data among land uses with AUML>10% (TS16 and TS17) and sites above and below ASML=3% is brought out in Figures 5.3a and 5.3b. Except for previously noted differences for elevated Fe at Federal Creek, magnitude of transformed %C, Cu, Zn, and Fe were not distinguishable among these groups either. For Mn, AUML>10% had significantly lower levels ($n=57$; $\alpha = 0.05$; [$P = 0.0022$]; $1-\beta = 0.92$) by parametric testing. One of the samples at TS16 showed nominal levels, reducing the difference between this group and average LogMn for ASML<3% to 0.32 g/kg. Paulins Hill Run (TS16) was the only 1st order stream studied. The site was apparently subject to greater temporal variation in AMD impact, observed also in biodiversity abundance data in Chapter 3. With parametric tests rejected for borderline unequal variance, non-parametric testing indicated significant differences between ratios of LogMn/LogFe for AUML>10% and ASML>3%, and between ASML<3% and ASML>3% ($n=57$; $\alpha = 0.05$; [$P = 0.0013$]; $1-\beta = 1.00$). With no adjustment made for the intercept in the linear relation, median LogMn/LogFe values were 0.69, 0.65, 0.63, and 0.52 for ASML>3%, Federal Creek, ASML<3%, and AUML>10%, respectively.

Historical sediment data from streams in the Western Allegheny Plateau Region from Ohio Environmental Protection Agency (OEPA) was also examined. Figures 5.4a and 5.4b combined data from two published studies on the Hocking River Basin and southeast tributaries of the Ohio River, respectively (OEPA, 1991a, 1991b). The latter study encompassed Leading Creek and the reference watershed Symmes Creek, and the former study included Federal Creek. Mn data was not made available in the 1990 OPEA studies. For comparison, reference lines from Figures 5.3a and 5.3b were included in Figures 5.4a and 5.4b, respectively.

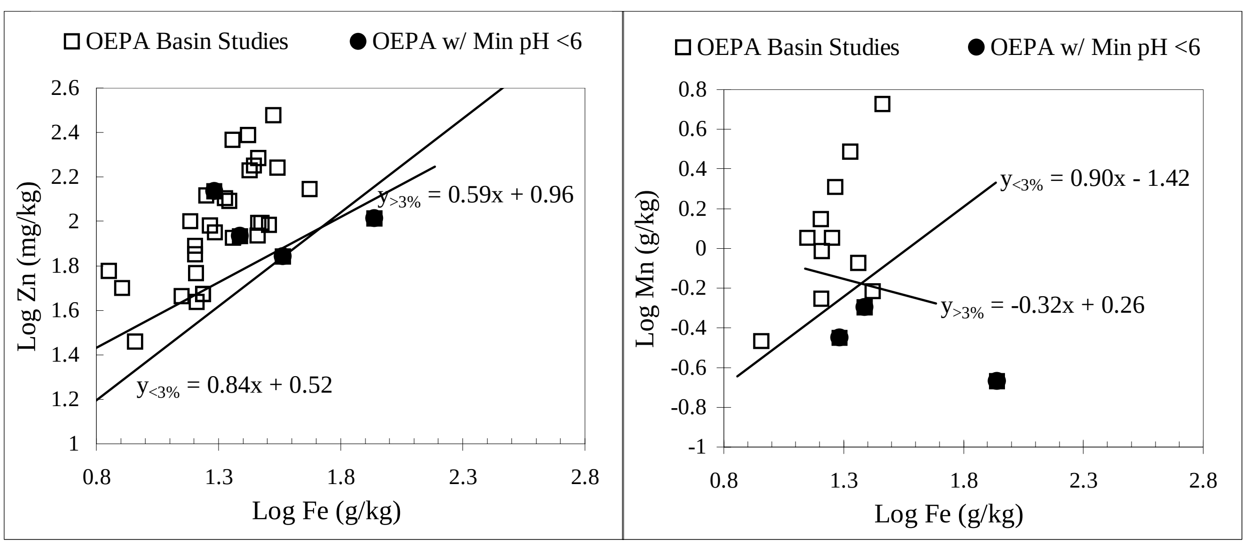

Figure 5.4a and 5.4b – (a) Zinc and (b) Manganese versus Iron Chemistry in Sediment – Southeastern Ohio STORET Station Results for 1990 to 1998

Due to greater variation observed in 1990 OEPA Zn/Fe ratios compared to our results collected in 1996 to 1997, additional data was obtained from STORET through OEPA. Available Fe and Zn sediment 1995 data was added. STORET was the source of all Mn data in Figure 5.4b. STORET data was not inclusive of all published 1990 Fe and Zn data, and included Mn data from 1990 and 1995. OEPA Methods for chemical analysis were reportedly similar, using USEPA Method 220.7 (ICP).

5.5 DISCUSSION

Non-elevated levels of Fe, Mn, Zn, and Cu were found in Leading Creek sediments compared to average crustal abundance shown in Table 5.4. Within Leading Creek, significantly higher Cu, Zn, Mn and Fe sediment concentrations at TS11 appeared to be related to the NPDES discharge in that tributary associated with active underground discharges. Sediment pollutant loading from Parker Run though was not discernable comparing Parker Run levels to all other tributary stations combined with upper mainstem stations above Parker Run. Sediment pollutant loading from Parker Run was also not evident in mainstem stations below Parker Run. This indicated that the effects of elevated metals in sediment were localized to the receiving tributary. TS11 contributes 8% of the drainage area at the nearest downstream mainstem station LCS6. More modest but overall higher levels of Fe were found at the reference site Federal Creek and few important distinctions were otherwise noted between land use/cover and sediment chemistry.

In addition to elevated metals associated with active mining, Fe and Zn rich solids were preferentially associated with small urban subsheds with high silt content, coal beds, and AMD sites with visible oxyhydroxide flocs. The observation for non-AMD sites with greater %fines, shown in Chapter 3, tied with constant sediment LogZn/LogFe ratios across the watershed, was consistent with the assertion that clay/silt size washload materials <0.125 mm were comparatively zinc and iron rich compared to courser grains, irrespective of land use. Mn in sediment and coal beds manifested a different character. The lowest levels of Mn were attributed to coal beds and sites with severe AMD. The highest levels were tied to the active mining process, noted previously. Greater variation in Leading Creek LogMn/LogFe sediment data was observed at sites with ASML>3%, with some but substantially less variation observed in LogZn/LogFe ratios. Consistent LogMn/LogFe sediment ratios were found for some AMD impacted site data and subsheds with strip-mines. Variation in soil and bedrock, as opposed to mining land uses may dominate the fluctuations observed in elevated LogMn/LogFe and slightly elevated LogZn/LogFe ratios for sites with ASML>3% (where AUML<2%). It is also feasible that a temporal water quality component plays a key role in Mn data, such as pH at AMD sites.

5.5.1 Coal Bed Chemistry and Sources of Instream Metals

High levels of Zn and low levels of Mn in proportion to Fe observed in ash content of coal beds in Leading Creek were highlighted in Figure 5.3a and 5.3b. Low Mn/Fe ratios were found in both shallow and deep coal beds mined in Leading Creek. On a whole ash basis levels of Zn and Mn were comparable to average crustal abundance and levels found throughout Leading Creek. Zn/Fe ratios were also roughly the same or, due to lower Zn, slightly lower for coal ash compared to sediment. Coal ash also had the overall highest levels of Zn and Fe found compared to all sediments. Magnitude and relative proportion of Zn and Fe iron in sediment influenced by treated AMD was remarkably similar to coal mined in that process. For active mining, data suggested that Zn concentrations followed source coal ash levels, and increased Mn by an order of magnitude. Coal preparation plant treatment of AMD involves bench materials and uses chemical oxidation for Fe, Zn, and Mn removal. This implied that clastic bench materials from the deep mines contained a significant source of Mn not correlated to iron. This supported a conclusion in Leading Creek that Zn and Fe source from fine materials (< 0.075 mm to 0.125 mm), and Mn sources from coarser sand and aggregate materials (Robbins *et al*, 1997).

For untreated AMD from AUML, Zn in sediment was average compared to all stream stations, and overall much lower than associated levels seen in the Redstone #8A coal. For subsheds heavily influenced by AMD, Mn in sediment and Mn in associated coal were similarly low relative to other Leading Creek stations. Thus for AUML>10%, low Mn in sediment was found along with low Mn in source coal ash. For AUML>10%, Zn in sediment was lower then in coal bed ash, and tracked the same Zn/Fe ratio across the watershed. The proportions of Fe, Zn, and Mn found in sediments and the water column indicated that the source of Zn instream could easily originate from finer inorganic minerals in close proximity to the coal beds. This was supported by a site-specific geologic setting in Leading Creek that indicated AUML coal seams were overlain by shale/clay, where they are also likely underlain by similar materials.

5.5.2 Sediment Metal Ratio Analysis

Coupled behavior of Zn, Mn, and Fe were evaluated through ratio analysis. With potential importance to watershed modeling efforts in AMD impacted regions, consistent Zn/Fe and Mn/Fe sediment concentration ratios were usually found at sites in Leading Creek without significant abandoned mining influences. Shown in Figure 5.3a, LogZn/LogFe was relatively consistent over a wide range of mining land use descriptions and ranges of pH in Leading Creek. Shown in Figure 5.4a, this did not hold true across widely dispersed watersheds in southeastern Ohio. Most other sites monitored by OEPA in a study of 13 southeast tributaries of the Ohio River, which included Leading Creek station LCS7, had higher LogZn/LogFe ratios attributed to higher zinc (OEPA, 1991b). Comparatively higher LogZn/LogFe ratios were also observed in another study in 1990 by OEPA of 8 watersheds in the Hocking River Basin (OEPA, 1991a).

Higher correlations among sediment data for Fe, Zn, and Mn were obtained by excluding sites with ASML>3%, seen in Table 5.5 and Figures 5.3a and 5.3b, but gradation with %ASML was not consistent in realizing elevated Zn or Mn with respect to Fe. Higher Mn/Fe or Zn/Fe ratios did not appear to be directly related to particle size analysis by examination of total %silt/clay at each site (all materials <0.075 mm in total materials <2 mm). For sediments with ASML>3%, overall higher soil acidity and sand content though were concluded to represent the source of higher Mn (Robbins et al, 1997). Discussed in Chapter 7, modest but significantly higher soluble Zn and Mn were found at sites with ASML>3% and AUML<2%. Attributed to low water column pH, TS17 and to a lessor degree TS16 both had lower Mn/Zn and Mn/Fe ratios than other sites, and had the highest levels of soluble Zn and Mn and the lowest pHs.

The two OEPA studies looked at sites with modest mining influences and a few sites with severe AMD. In Leading Creek, more subtle departures from Zn/Fe trends at some sites in the mining belt were observed with ASML>3%. This was attributed to disturbed soil type and not necessarily mining activity. It may be that due to variation in geology disturbed by a variety of land uses, southeastern Ohio watersheds can exhibit different Fe, Zn, and Mn metal ratios than the dominant trend found in most Leading Creek subsheds not mined. OEPA's sample results for Leading Creek could also point to differences in sampling and analysis that might invalidate this conclusion, and allow acceptance of a more widespread trend in Zn/Fe ratios throughout the ecoregion. In assessing Leading Creek study reference controls, OEPA data for Symmes and Federal Creek exhibited Zn/Fe ratios similar to data collected here. Non-supportive of data collected here for station LCS7, Zn was 300% to 650% higher in OEPA's results from 1990.

In Leading Creek, it was shown in Figure 5.3b that Mn/Fe ratios were much more variable at sites influenced by abandoned mining, particularly at sites with low pH. Mn/Fe ratios were not correlated though to aereal mining extent or level of stream acidification. Supporting this conclusion, for sites downstream of soil series associated with surface mining, unusually high Mn/Fe ratios were not associated with typical markers of AMD impact to ecology such as poor benthic macroinvertebrate biodiversity. Low Mn/Fe ratios were found at the two sites having the lowest biodiversity scores and lowest pH which represent two of the best measures available for estimating ecological impact attributable to AMD (Allan, 1995; USEPA, 1980).

More consistent Fe and Zn sedimentation compared to Fe and Mn was attributed in part to relatively slow biotic and autocatalytic abiotic oxidation of Mn(II) for acidic and natural ranges of pH (Cornwell, 1990; Diz, 1997). In summary of all sediment chemistry data examined in Leading Creek and other southeastern Ohio watersheds, low Mn/Fe was a sufficient indicator of severe AMD and poor biodiversity, but not a consistently necessary condition. Mn oxide coatings may be dissolved from instream sandstones (Robbins *et al*, 1997). Fluctuations might be tied to hydraulics and temporally distributed flooding and scour, as well as lower regime transport of fine sands which could affect residence time at the sediment-water column interface.

5.5.2.1 Historical Sediment Studies in Southeastern Ohio Watersheds

Sediment chemistry data from OEPA at a station just below (R)SCS2 on Symmes Creek at river kilometer 17 indicated relatively low iron at 14,000 ppm and zinc at 46.2 ppm (OEPA, 1991b), falling close to trends for Fe/Zn in Figure 5.4a. However, most other stations in OEPA's study of southeast tributaries of the Ohio River subject to impact from various mining activities exhibited characteristically higher Zn/Fe ratios, shown in Figure 5.4a. In direct comparison, 1990 sample data at OEPA's reference site on Leading Creek equivalent to station LCS7 studied here (ASML = 1.1%) exhibited iron levels of 22,100 ppm versus 11,000 and 17,100 for 1996, and 1997, respectively. Somewhat at odds with results here though, driving the higher ratio found in 1990 at LCS7 and other OEPA study sites, elevated zinc of 124 ppm was measured by OEPA versus 19 ppm and 41 ppm measured at LCS7 in 1996 and 1997, respectively. Except for one site on Meadow Run, average pH for the OEPA study sites outside of Leading Creek were between 6.9 and 7.9 with the lowest minimum pH of 6.0 (OEPA, 1991b). Meadow Run had pH values between 4.2 and 7.3 with average pH 5.3, most similar to Thomas Fork. This Meadow Run site had a Zn/Fe ratio falling directly on Leading Creek's trend with levels of Fe and Zn of 36,600 ppm and 69.5 ppm, respectively, also closest to values at TS17.

Evaluation of a similar study conducted by OEPA in the Hocking River Basin (OEPA, 1991a) except for one site indicated relatively high Zn/Fe ratios. Though OEPA did not sample sediment at Federal Creek residing in the same basin, this site had Fe and Zn levels in this study of 42,900 ppm and 75.4 ppm, respectively, falling directly on trend. The one exception for the OEPA Hocking River study showing a Zn/Fe ratio similar to values found in Leading Creek was that of heavily AMD impacted Monday Creek. Monday Creek had Fe and Zn levels of 86,900 ppm and 103 ppm, respectively, and average pH 3.9 (n=5, $\sigma = 0.64$). Also heavily damaged by AMD with average pH 4.1 (n=4, $\sigma = 1.4$), nearby Sunday Creek fell considerably off the Leading Creek trend line with Zn levels of 137 ppm, and 19,200 ppm Fe. In Meadow Run, one sample for Mn/Fe fell on the trend for Leading Creek, and one was considerable above it. A similar site pattern was observed for Mn data on Sunday Creek were Mn/Fe was above trend, and Monday Creek represented the unusually low value falling off the Mn/Fe trendline.

Differences in Zn/Fe may simply reflect variation in geology and AMD equilibrium across the watersheds examined. Sampling and analytical methods between this study and OEPA's work may though explain some or all variation observed, but these facets could not be well defined with available information. ICP tests methods utilized in our study and in the OPEA studies were reportedly similar (personal communication with OEPA lab manager Pat Smith, May 2000). Excluding natural variation in geology, range of calibration of respective ICP analyzers would be suspected as the most likely potential source of variation between OEPA studies and 1996-1997 results, where contamination for Zn might also be a suspected effect.

5.5.2.2 Active Mine No. 31 Dewatering Event

Not shown in Figures 5.3 or 5.4, 1994 sediment data gathered as part of a risk assessment associated with dewatering AMD from Mine No. 31 in 1993 was also evaluated (Birge *et al*, 1995). In 1993, a billion gallons of AMD was pumped from Mine No. 31 with significant portions discharged untreated to Leading Creek. Analysis of Birge's (1995) sediment data associated with post-dewatering assessment of Leading Creek in August 1994, 1-year later, indicated a supporting relation for $LogZn = 0.71LogFe + 0.82$ ($R^2 = 0.74$) using EPA Method 3050 (USEPA, 1986) for extraction. A relation was also found for $LogZn = 1.0LogFe + 0.38$ ($R^2 = 0.60$) using ASTM Method D 3974-81 (ASTM, 1989) for extraction. Atomic absorption spectrophotometry (AAS) with flame atomization was used for analysis of these samples. Sample extractions employed HNO_3, H_2O_2, and HCl in the digestion. Birge's samples spanned the lower mainstem below LCS4 and several stations on Parker Run. Also included were several samples just outside of Leading Creek in Raccoon Creek that also received untreated AMD.

To varying degree, many of the sites sampled by Birge represented multiple land uses, including AUML and ASML, with most having the potential for AMD impact associated with the dewatering event. Birge's LogZn/LogFe sediment results indicated consistency with the curve shown in Figure 5.3a, with LogZn/LogFe again apparently invariant to multiple land uses. There was, however, poor recovery observed between the two digestion methods (<50%) seen in Fe data < 30,000 ppm that trended with poor recovery in Zn data < 50 ppm. For Mn data > 750 ppm, similarly poor recovery was exhibited and therefore formal assessment of Mn/Fe was not attempted. Similar strong LogMn/LogFe ratio patterns shown in Figure 5.3b were not consistently discernable. Some sites near Mine No. 31 and sites below Thomas Fork TS17 did indicate ultra low Mn and departure from an increasing trend mimicking Figure 5.3b.

5.5.3 Sediment Quality and Soil Distribution for ASML>3%

Directly comparing regression equations in Figures 5.3a and 5.3b, significant differences between equations for Fe and Mn, and again for Fe and Zn results were observed for sites above and below a criteria of ASML=3%. Spatially distributed soil type as opposed to mining activity, however, appeared to be the underlying determinant in this trend. Given unremarkable sediment concentration levels compared to crustal abundance data, dominance by soil type and associated hydrogeologic characteristics such as thickness, permeability, and acidity might be expected (USEPA, 1980). Stations LCS9 and TS14 showed the greatest departure from the consistent ASML<3% trend where Station LCS8, which lies between these two sites, fell right on trend.

Seen in Figures 5.3a and 5.3b and Table 5.5, sediment-soil relationships may be to varying degree complicated by temporally and spatially distributed upstream acidification and associated AMD equilibrium chemistry. Discussed in Chapter 7, significant differences were observed in slightly higher soluble Zn and Mn in subsheds with ASML>3% for low and medium flows, compared to ASML<3%. For all stations except TS15, TS16, and TS17, average pH ranged 7.3 to 9.1, where TS15 had average pH 6.7. ANOVA results indicated overall no differences between median pH for sites with ASML<3% and sites with ASML>3%. Thus both ASML>3% and ASML<3% appeared to be well buffered with respect to inflow acidity.

Because the ASML mining belt in Leading Creek is quite distinguishable along a transition line of major soil associations (scale 1:100,000-250,000), individual soil series within the Upshur-Gilpin-Pinegrove mining belt soil association (Gillmore *et al*, 1991) might account for relationships between Zn, Mn, and Fe at stations without severe stream acidification. Dominant soil series in this association tend to be overall more acidic and characterize the strip mining belt given by ASML>3%. A second stream acidification process of acid rain is known to occur in this region (USEPA, 1980). Impact of acid rain would be expected in Leading Creek to be dominated by secondary influences such as soil type, characteristics, and vegetative cover. Thus two minor acidification processes are constantly at work. Compared to AUML>2% these were negligible with respect to AMD. While it is unknown if the two mechanisms contribute to higher Mn/Fe and Zn/Fe, average Mn and Zn were higher in sediment and water columns.

Actual mining activity may not be a sole determinant in higher than normal Mn/Fe sediment chemistry ratios in Leading Creek. Two subsection data points for LCS2 in 1997 having normal Zn/Fe ratios exhibited similarly high Mn/Fe ratios as the group defined by ASML>3%. No coal mining was known to have occurred upstream of LCS2 which had the best overall levels of biodiversity. Upland geomorphology of the nearest tributary, Five Mile Run (TS3B), was dominated by similar soil series in the Upshur-Gilpin-Pinegrove association. TS3B was also heavily influenced by agriculture activities, and, with thinner Monongahela Group bedrock, was not mined. Thus simple disturbance of upper horizon ASML soils could be a causal factor linked to high Mn/Fe and Zn/Fe. Station TS3B had the fourth highest average level of Mn in the water column next to TS15, TS16, and TS17. TS15 had normal Mn/Fe ratios for both years. In the absence of instream acidification, higher soluble Mn appeared to lead to higher levels of Mn in sediment. Excluding severe AMD, subtle departures in Leading Creek from the trends observed in Zn/Fe and more variable changes in Mn/Fe sediment concentration ratios in Figures 5.3a and 5.3b were not apparently due to strip mining land uses only.

5.6 CONCLUSIONS

Key trends and patterns in the relation of AMD water quality pollutants were identified in chemistry of streambed sediment and disturbed coal beds. In Leading Creek active mining exhibited modest, but significantly higher Fe, Mn, Cu, and Zn compared to other land uses. For all land uses though levels in sediment and coal bed ash were within range of average crustal abundance for each parameter, with slightly higher Fe in coal beds. Inter-annual variation in %C and Cu were highest, and generally weakly related to Fe, Zn, and Mn, the latter strongly correlated in sediment. Carbon and copper sediment concentrations varied the most in smaller subsheds dominated by agricultural activities. Tracking the highest levels of %silt/clay, small urban subsheds < 10 km^2 exhibited the highest levels of carbon in sediment at >1%. High carbon levels >0.6% to 0.8% were associated with small urban subsheds <20 km^2, but this aereal threshold criteria was not a significant indicator across all subsheds in Leading Creek.

In Leading Creek strong trends were observed in sediment between Zn and Fe where LogZn/LogFe ratios were relatively constant across all land uses, with some variation observed for ASML>3%. Highest Fe and Zn were found in coal bed ash, representing pyretic clay/shales in close proximity to coal beds where higher iron was observed in coal compared to sediments. Significant differences were found between ratios of LogMn/LogFe for AUML>10% and ASML>3%, and between ASML<3% and ASML>3%. A paucity of Mn was often observed in coal beds and stream sediments with average water column pH<6. In contrast, a higher than expected ratio of LogMn/LogFe was sometimes found at sites with ASML>3%. Sandstones appeared to represent the source of Mn in soils and sediments where instream dissolution was attributed to the cause of Mn paucity. Sandstones were also concluded to represent the source of higher Mn in sediments with ASML>3%, where modest but significantly higher soluble Zn and Mn were can be found with no signs of instream acidification. Strip mine activity was not unique in the expression of higher Mn/Fe ratios, where similar soil series disturbed by only agricultural activity exhibited this distinction. Outside of Leading Creek, greater variation was observed in Zn/Fe and Mn/Fe trends based on historical OEPA studies of southeastern Ohio watersheds. Those differences might be explained by variation in geology or analytical methods.

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Chapter 6 - TREATED AND UNTREATED ACID MINE DRAINAGE WATER COLUMN SIGNATURES IN AN OHIO WATERSHED WITH ACTIVE AND ABANDONED COAL MINING LAND USES

Justin Eric Babendreier

(ABSTRACT)

A 388 km^2 watershed heavily impacted by various mining techniques and acid mine drainage (AMD) was studied. Historical abandoned mining in Leading Creek involved both underground near-surface mining (AUML) and surface contour methods commonly referred to as strip mines (ASML). Active deep long wall mining operations are conducted in the watershed today and discharge significant loads of treated pH-neutralized AMD. Total water column Mg, Zn, Fe, and Mn were studied along with pH, non-filterable residue (TSS), and sulfate. In addition the metal product $[Mn \bullet Zn]^{0.5}$ and Zn/Fe ratio were considered. The analysis addressed two key hydrologic-hydraulic factors including flow regime and %AUML normalized to drainage area. AUML was quantified using threshold criteria for AUML=2% and AUML=10%. Site-specific flowrates were determined for all water quality data. Water quality responses were segregated using threshold criteria normalized to drainage area defining drought, low, medium, and high flow conditions, respectively.

Looking at active and abandoned coal mining, untreated AMD from AUML>10%, and to a lessor degree AUML>2% dominated water quality concerns associated with low pH and elevated Mn and Zn. For a given level of AUML, predictable, quantified gradients in flow regime imparted elevated AMD derived soluble metal concentrations during increasingly dryer conditions associated with baseflow discharges. High levels of dissolved Mn can be good predictors of AMD, but not during drought. For low and medium flows, total Zn and Mn were determined to be representative of soluble species, where suspended Zn and Fe rich flocs associated with fine materials <0.125 mm can increase in proportion during higher flow conditions. Tying in with previous work on sediment ratios, the conclusion reached was that total Zn/Fe ratios in sediment were non-differentiable with respect to coal mining, and total soluble Zn/Fe could better identify conditions with elevated soluble Zn.

Total Mn was primarily of soluble form during low and medium flows, and increased modestly in suspended form during high flow monitoring of washload. Suspended Mn in washload data would be expected to be associated with suspensions of coarser sand fractions in sediment. Based on analysis of water column mechanisms, paucity of Mn in sediments at some sites with severe AMD was attributed to *in situ* dissolution of sandstone materials in the sediment bed. Magnesium and sulfate signatures within Leading Creek were strongly segregated among limestone quarry and coal mining land uses excluding strip mines. Unique linear relationships between Mg and sulfate were found for limestone quarry operations without AMD impact, for treated AMD without untreated AMD impact, and for untreated AMD without treated AMD impact. Zn and sulfate were strongly correlated for untreated AMD derived from AUML.

Chapter 6 - TREATED AND UNTREATED ACID MINE DRAINAGE WATER COLUMN SIGNATURES IN AN OHIO WATERSHED WITH ACTIVE AND ABANDONED COAL MINING LAND USES

6.1 INTRODUCTION

Many eastern United States Appalachian streams have been impacted by coal mining activities. In these landscapes both point and nonpoint sources of pollution play important roles in the decline of aquatic freshwater ecology. Defining risk on watershed scales requires increased knowledge of major processes and environmental mechanisms associated with active mining, abandoned mined lands (AML), and their respective pyretic acid mine drainages (AMD). Despite AML and AMD's profound impacts upon water quality, only a limited understanding of the interconnections between source pyrite chemical quality, commingled mining land uses, and resulting instream water and sediment chemistry have been described in literature. This is due to extreme variation of controlling factors that occur on the landscape.

Underlying hydrologic-hydraulic macro-mechanisms (e.g. disturbance, oxidation, and discharge) play pivotal roles in determining how AMD manifests as stress to aquatic organisms. To further understand AMD's impact, instream signatures of upland pyrite oxidation and acid generation need to be better assimilated into ecological risk assessment frameworks for complex ecosystems like watersheds. In this paper, examination of AMD expression instream related to both sediment and water column chemistry was conducted as part of a larger hydrologic-based ecological risk assessment of Leading Creek, located in southeast Ohio.

6.1.1 Formation of Acid Mine Drainage in Watersheds

Detailed knowledge of sulfide mineral oxidation was described in literature decades ago along with the impacts of AMD attributed to mining hard rock ores (e.g. copper, gold, etc.) and coal (Dunne *et al*, 1978; Leopold *et al*, 1964; Nordstrom, 1982; Singer *et al*, 1970; Snoeyink *et al*, 1980). These earlier attempts are today still often referenced frameworks for describing the reactions resulting in acid mine drainage. An evolving issue, researchers began to recognize the important roles played by *Thiobacillus ferrooxidans* and other iron eating bacteria such as *Thiobacillus thiooxidans* and *Ferrobacillus ferrooxidans* (Daniels *et al*, 1995; Evangelou, 1995; Singer *et al*, 1970). The acidophilic, mesophilic, microaerophilic bacteria represent rate limiting determinants in AMD kinetics in natural settings. An obligate chemoautotrophic aerobe, *T. ferrooxidans* dominates this process and has optimum growth in the range of pH 1.5 to 3.5 (Diz, 1997; Kuenen *et al*., 1992). Optimized relative activity of *T. ferrooxidans* has been reported near pH 3, 30 ^{o}C, and at O_2 molar fractions = 1% (Evangelou, 1995; Jaynes *et al*, 1984). Cell activity was reported as invariant to O_2 above 1% where pH and temperature represented local maxima. No significant growth of *T. ferrooxidans* occurs in saturated environments where O_2 diffusion is 10^{-4} times slower than diffusion in unsaturated waste pile systems (Evangelou, 1995).

Two basic processes are involved in AMD generation including abiotic chemical oxidation and microbiologically mediated oxidation of sulfide minerals (Lowson, 1982; McKibbon *et al*, 1986; Nordstrom 1982; Singer *et al*, 1970; Snoeyink *et al*, 1980; Williamson *et* *al*, 1994). These are thought to include several types of redox reactions, hydrolysis, complex ion formation, and solubility controls (Nordstrom, 1982). Most researchers continue to recognize the basic system of biologically mediated chemical reactions as one that it is extremely complex, both thermodynamically and from the standpoint of kinetics (Williamson *et al*, 1994; Evangelou, 1995). This is more true in AMD's expression in natural environs where depositional environment, paleoclimate, modern climate, geology, hydrology, and man vastly complicate AMD expression (Caruccio *et al*; 1974; Cecil *et al*, 1985; Curtis *et al*; 1973, 1979; Dickens *et al*, 1986, 1989; Leopold *et al*, 1964; Nordstrom, 1982; Schumm, 1977; Skousen, 1996).

Summarized by Evangelou (1995), three interdependent approaches can be used to study pyrite oxidation: a) inorganic/biochemical reactions, b) electrochemical reactions, and c) reaction kinetics. None are sufficient in and of themselves. Two constructs can be taken to conceptualize and describe underlying reactions and rates that govern sulfide oxidation and acid mine drainage. One is considered a classical overview interpretation of pyretic acid mine drainage generation (Nordstrom, 1982; Snoeyink *et al*, 1980). This stoichiometric approach though does not address surface chemistry phenomena. Differences among researchers can be found regarding the importance of relative roles of indirect and direct metabolic oxidation and the interrelated role of surface chemistry (Evangelou, 1995). More recently it has been suggested that the interaction of dissolved oxidants and the pyrite surface is not actually site-specific, where anodic and cathodic reactions may occur at different locations on the mineral surface (Williamson *et al*, 1994).

6.1.1.1 Classical Description of AMD Reactions and Pollutants

Researchers appear to agree on the overall stoichiometric reactions taking place and the ascribed to fact that significant rates of oxidation reactions are biologically controlled in natural environments. Microbial oxidation of Fe(II) by ferrooxidans can be over 10^6 times faster than chemical oxidation (Singer *et al*, 1970; Snoeyink *et al*, 1980) and oxidation rates are less sensitive to changes in pH below 6 s.u. (personal communication with D. Rimstidt, 1997). Evangelou summarized other researcher's observations that reduction in pyrite oxidation above pH 4.5 can be observed but accelerates again in neutral to alkaline waters. Overall, observations appear to indicate that across acidic pH ranges, the Fe (II) oxidation rate by *T. ferrooxidans* is relatively stable below pH 6, and dominates other mechanisms available to oxidize pyrite.

The classical description of biologically mediated chemical reactions related to pyrite oxidation is summarized in Figure 6.1. Oxidation of pyrite mobilizes reduced sulfide metals and generates acid, causing dissolution of bedrock and soil materials in water flow pathways. Precipitation of oxyhydroxides will typically follow oxidation of metal sulfides, involving a second stage of acid generation (Evangelou, 1995; Nordstrom, 1982). According to Nordstrom the sulfide mineral source can include various sulfide compounds of Fe, Mn, Cu/Fe, and As/Fe, and precipitation can include other oxyhydroxides, sulfate and chloride salts. Shown in Figure 6.1 elevated mass concentrations will typically be observed for iron, manganese, and aluminum, and several heavy metals including zinc, nickel, and arsenic among others.

Mechanisms summarized in Figure 6.1 are presented for aid in interpreting environmental effects of the AMD generation processes seen instream. AMD processes can generate high levels of water-soluble sulfate up to 12,000 mg/L, low pHs ranging down to and below 2 s.u., total hydrogen ion acidity up to more than 20 mg/L with mineral acidity reaching levels of 1000 mg/l or more, and ferrous iron levels ranging up to 500 mg/L or more (Bigham *et al*, 1996; Caruccio *et al*, 1974; Evangelou, 1995; Hedin *et al*, 1994; Nordstrom, 1982; Skousen, 1996; Snoeyink *et al*, 1980; USEPA, 1976, 1980). Elevated iron, manganese, and aluminum concentrations up to 440 mg/l, 125 mg/L, and 270 mg/L, respectively, were noted in surveys of eastern coal mining operation wastewaters associated with treated AMD (Diz, 1997; USEPA, 1976). In contrast to untreated AMD, total acid rain acidity entails little mineral acidity and ranged between 25 to 100 mg/L in studies in West Virginia (Skousen, 1996; USEPA, 1980). Major mechanisms and impacts of AMD from coal mining are further summarized in Chapter 5.

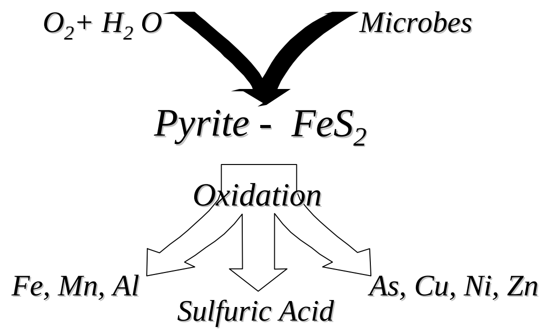

Figure 6.1 – Overview of Acid Mine Drainage Generation

6.1.2 Instream AMD Chemistry Study Objectives

Attempts to further relate knowledge of AMD water and sediment chemistry in natural, complex settings were directly investigated here for potential future applications in development of risk assessment frameworks. This was attempted through identification of key *in situ* trends in mineral and metal partitioning in stream sediment and the water column. The hypotheses tested here addressed aspects of hydrologic-based watershed-scale ecological risk assessment by employing GIS based land use/cover analysis. The underlying objective was to evaluate quantitative descriptions of urban, agriculture, and surface and underground coal mining, and to relate their impacts to sediment and water chemistry. Few studies of commingled active and abandoned mining exist that combine Geographical Information System (GIS) at fine scales of 1:24000, land use/cover analysis, and risk assessment.

Work reported here was developed in testing the hypothesis that unique instream water column signatures for agriculture, quarry operations, active underground mining and AML land uses, respectively, can be developed from water quality concentration data, accompanying flow data, and drainage area. This paper completes a discussion on sediment quality processes introduced in Chapter 5 and their relationship to pollutant loading from the landscape. The hypothesis was intended to evaluate relationships between land use and water column quality and toxicity, and land use and pollutant loading rates, respectively. Breaking down complexity of the hypothesis, critical supporting aspects were first investigated in this paper by comparing land use/cover variables to measures of sediment and water quality, focusing on how contaminant phases are related for key AMD metals. Identification of fundamentally sound, reliable signatures of AMD instream were sought. A more fully developed risk assessment of coal mining land uses and flow regime and their effects on water quality are presented in Chapter

7. The hypothesis will require further reporting on pollutant loading rates to complete the discussion, particularly for demonstration of water column signatures for non-mining land uses.

6.2 STUDY WATERSHEDS

Leading Creek is a fifth order stream (scale: 1:24000) 96% of which spans Meigs County, with other portions in Athens, and Gallia Counties. Together, 11 mainstem stations and 19 tributary stations shown in Figure 5.1 in Chapter 5 were selected for hydrologic, hydraulic, and ecological risk assessment monitoring. Mainstem stations were located to provide complete longitudinal coverage of Leading Creek, and discrimination among potential sources of sediment or contaminant impacts to the stream system. Water quality monitoring activities also included three reference sites located in Symmes Creek and Federal Creek. Leading Creek and the two reference watersheds along with all southeastern Ohio streams reside in the Western Allegheny Plateau ecoregion, one of four ecoregions covering the state (Hughes *et al*, 1994; Omernik, 1987; OEPA, 1991a, 1991b; USEPA, 1988; Yoder *et al*, 1996). Leading Creek and reference site monitoring stations were described in detail in Chapters 3, 4, and 5. Of interest in water quality assessment, Parker Run (TS11) and Ogden Run (TS6) in Figure 5.1 receive active coal mining treatment plant effluents and assorted stormwater and sanitary discharges. Other NPDES sites in Leading Creek relate to sanitary treatment (STP), water treatment (WTP), and quarry discharges.

6.3 METHODS AND MATERIALS

Data collection in Leading Creek and the reference watersheds included monitoring of: (1) Land use and land cover, (2) precipitation, stream stage, water velocity, and flow; (3) sediment and water quality; (4) benthic macroinvertebrate and stream habitat; and (5) toxicity through sediment, acute, and *in situ* testing programs. To assess chemical, physical, and biological interactions, stations in Figure 5.1 were examined over a wide range of drainage areas and stream orders. Data collection methods specific to measurement of hydraulics, water quality, and land use/cover are outlined or detailed here.

6.3.1 Land Use and Land Cover

Spatial analysis of subsheds used several geodata sources including data hand-digitized from USGS quadrangle 7.5-minute map series and USGS hydrology geodata obtained from Ohio Division of Natural Resources (ODNR). Land use/cover data (scale 1:24000) for coal mining land uses was developed through two primary sources including a) 7.5-minute quadrangle series maps depicting abandoned strip mines, b) Ohio Division of Natural Resource (ODNR) geodata depicting abandoned underground mines. Drainage areas describing monitored tributaries span approximately 124 to 8038 ha, and are shown in Table A1 in Appendix A.

6.3.2 Stream Hydraulics

Equipment at each of two continuous gage stations (CGS) included an ISCO Model-4230 bubbler level meter and Model-675 automatically tipping rainfall gage. Continuous monitoring with automated data storage allowed comparison of stream stage (+/- 0.025 cm), flow (+/- 10 to 20%; estimated), and rainfall data (+/- 0.025 cm). Flow data accuracy is site dependent and regime dependent. Stage at each of 16 manual gage sites was usually also monitored during water quality sampling. Southern Ohio Coal Company (SOCCO) staff collected weekly stage readings and daily rainfall at their Mine No.2 Coal Office at Point Rock, Ohio. Manual staff gage control points (inverted guides) were constructed of 2.5 cm diameter PVC tubes, approximately 0.5 to 1.5 m in length, rigidly attached to a bridge guardrail. Use was also made of a Solinst-100 electronic water level meter with audio signal or a steel tape measure together with line-of-sight to the tape end and water surface, both with similar accuracy (+/- 0.03 cm).

Flowrates were determined through standard methods employed for stage-discharge rating development at each gage site. For assessment of water quality data, event-day flowrate models were used here to estimate flowrates at ungaged sites and sites with missing data. For assignment of flow regime in water quality data, for each site and for each event site-specific flowrate was used. Flow regime and flowrate modeling methodologies used are detailed in Appendices C and D respectively, with supporting hydraulic data in Appendices A, E, F, and G.

6.3.3 Water Chemistry

Water quality sampling was conducted at all mainstem, tributary, and reference sites shown in Figure 5.1 in Chapter 5 except station LCS5A used for supplementary biological data collections only. Monthly water column grab samples were collected in addition to periodic stormwater grab sampling including collection of trip and field blanks (APHA, 1992). Grab sampling at each stream site was conducted using a clean plastic bucket tethered to a rope and dropped from bridge sites into the swifter areas of flow. The bucket was rinsed with site water prior to use each time. Suspended solids collected at the stream surface represented sediment washload and would be expected to have negligible fractions >0.125 mm in particle size diameter. Unfiltered water quality samples were kept cool (<4 ^{o}C) and transported to American Electric Power's Dalton laboratory in Columbus, Ohio for analysis of pollutants indicative of urban, agriculture, and active and abandoned mining influences. The Dalton lab facility is used by AEP to meet NPDES permit compliance monitoring requirements for their regional fuel extraction and power generation facilities.

Sample analyses were completed within method prescribed holding times and included testing for sulfate, total suspended solids (TSS), and total Mg, Fe, Mn, Zn, and Cu among other parameters. Field measurements for water column pH, specific conductivity, and temperature were also completed. New 1L plastic collapsible containers were used and appropriate samples field preserved using nitric acid. Total metals in water quality samples were analyzed with inductively coupled plasma (ICP) via USEPA Method 220.7. Standard water and wastewater methods for other parameters were also used including USEPA Method 375.4 for determination of sulfate by turbidimetric analysis and USEPA Method 160.2 for gravimetric analysis of total non-filterable residue (TSS) (APHA, 1992). Method detection limits (MDLs) and practical quantitation limits (PQLs) for ICP metals (40 CFR 136), and limits of quantitation (LOQ) reported by AEP are summarized in Table 6.1. For interpretation of significance, this work assumed PQLs = max{2* MDL, LOQ} as being representative of actual ICP capabilities.

Not exclusive for all data collected, field meters used included Corning Model-90 Checkmate multimeters, an Orion Model-122 conductivity meter, a YSI-55 oxygen meter, and also a Fisher Accument Model-1003 pH meter. Specific conductivity was normalized to 25 oC (1.91% standard temperature coefficient; +/- 1 umho/cm) and water temperature (+/- 0.01 ^{o}C). Portable pH probes were calibrated with pH 4, 7, and 10 lab-grade buffers and conductivity was calibrated with standardized KCl solutions using lab-grade deionized water. Supplementing site-specific data, equipment at each continuous gage station included a YSI-600 water quality probe installed instream for easy retrieval. Continuous monitoring with automated data storage allowed comparison of temperature corrected pH (+/- 0.01 s.u.), specific conductivity normalized to 25 oC (1.91% temperature coefficient; +/- 1 umho/cm), and water temperature (+/- 0.01 oC). During routine maintenance visits at CGS stations, conductivity was calibrated with fresh 0.005 N KCl solutions, and pH 4 and 7 buffers were used (APHA, 1992). Stage at gage sites was routinely monitored during sampling events.

Table 6.1 – Analytical Methods for Total Metals and Detection Limits

Chemical	ICP MDL (ppb)	ICP Est. PQL (ppb)	AEP LOQ (ppb)
Copper	6	12	1
Iron	7	14	10
Manganese	2	4	10
Magnesium	30	60	100
Zinc	2	4	4
SO_4	-	-	1000
TSS	-	-	1000

Additional 1990 through 1998 water quality data for southeastern Ohio watersheds was taken from STORET, which included the dewatering period in 1993 associated with the release of 1 billion gallons of untreated AMD from Mine No. 31 due to a failure of a closed mine seal. This data was provided by Ohio EPA and included similar watersheds analyzed in Chapter 5.

6.4 RESULTS

6.4.1 Land Use and Land Cover Analysis

Land use/cover variables addressing active, ASML and AUML coal mining activities were summarized in Table 5.1 in Chapter 5 and can be viewed graphically in Figures 3.2 through 3.7 in Chapter 3. Characteristics and qualifications concerning geodata used here are described in more detail in Chapter 3 and Appendix A. Leading Creek coverage of abandoned underground mines in Table 5.1 represents 124 non-overlapping coal mine works where all reside above stream drainage elevations. In Figure 3.7, long wall mining operation aereal extents shown (>100 m BLS) at Mines Nos. 2 and 31 cover 6300 ha and represent deep groundwater diversion zones associated with treated AMD discharged into Leading Creek. Excluding historic and current SOCCO operations, most underground mining in the Leading Creek vicinity was conducted prior to 1910, primarily during the latter portions of the 19th century. Most strip mining occurred after WWII where 80% or 1379 ha of strip mines in Leading Creek were created by 1961, and 20% or 353 ha between 1961 and 1978. Little surface mining occurred after this period due to burdens imposed by the Surface Mining and Reclamation Act of 1977.

6.4.2 Water Quality Data and Stream Flow Condition

Average water quality data discussed here is summarized in Table I1.a and I1.b in Appendix I for each site. Water quality metal sample results discussed here were not filtered and n = 15 to 16, for indicators n = 17 to 20; and for reference sites n ≥ 5. These data represent the same sampling events shown in Figures 5.2a and 5.2b in Chapter 5 except water quality indicators were also collected during additional low flow sampling events conducted during late spring and summer 1997. For these added events, TSS data was collected in acid washed 1L plastic bottles. Average water quality data referenced incorporated all flow regimes (low, medium, and high). Water quality indicator data was summarized graphically in Chapter 5 and pointed out differences between pH and specific conductivity for various land use/cover categories. Sediment quality data and analysis discussed here was also taken from Chapter 5.

In comparison to active underground mining, minor point source permitted National Pollutant Discharge Elimination (NPDES) discharges given by the WTP, STP in Figure 5.1 in Chapter 5 and sanitary wastewaters from the active mines represent insignificant loading of treated effluents and flow into the stream system. The active mining operations must meet categorical effluent limitations established for coal treatment plants (USEPA, 1976; 40 CFR 434). Treatment of coal preparation wastewater involves primarily physical separation processes and pH neutralization using sodium hydroxide. Waste sludge results from precipitates of primarily iron, manganese, and aluminum oxyhydroxides and co-precipitation of heavy metals (Diz, 1997; USEPA, 1976). Coal preparation plant wastewater effluent is high in total dissolved solids (TDS) primarily composed of sulfate, sodium, and chloride where conductivity at SOCCO Mine No.31 typically ranges between 4.0 to 6.2 mmhos/cm.

During lower flow conditions the watershed can gain significant alkalinity from deep long wall mining mine operations, helping buffer AML-AMD entering the lower mainstem. The quarry discharge is also a significant contributor of headwater alkalinity during low and medium flows. For most periods of flow, Mine No.2 wastewaters are conveyed to Mine No. 31 for treatment and storage before final discharge to Parker Run (TS11). Following periods of increased groundwater infiltration after run-off, some process wastewaters are treated and discharged to Ogden Run (TS6).

Detailed hydrologic-hydraulic data characterizing each sampling event is summarized in Table D2 in Appendix D, along with event-day flowrate models used for estimation of instantaneous flowrates at ungaged sites and sites with missing stage data. The data there are instructive in communicating the wide range of hydraulic conditions incorporated into flow modeling and water quality data sets used in this analysis. Conditions monitored spanned both drought and the approximate 100-year flood recurrence interval. For flow regime determination, low and medium flow conditions were based on a separation threshold of $2.5x10^{-3}$ $m^3/s/km^2$, and medium and high flow conditions were separated at $1.1x10^{-2}$ $m^3/s/km^2$. Drought was defined as $<1x10^{-3}$ $m^3/s/km^2$. The justification for these criteria were detailed in Appendix C, and represent distinctions between primarily baseflow discharge conditions and stream flow dominated by runoff, where medium flow conditions transition between baseflow and runoff. Flow condition at station TS11 is qualified within this framework since almost all conditions *in situ* represent high flow due to NPDES discharges of diverted groundwater from Mine No.31's polishing reservoir. The 1996 water-year mean daily flow at gage CGS1/LCS5B was $1.5x10^{-2}$ $m^3/s/km^2$.

Detection limits in Table 6.1 do not address serial dilutions. Throughout the course of water quality monitoring for this study, 3 water quality samples collected at stations LCS8, LCS9, and TS11 were subject to serial dilution during metals analysis for the second day of the August 1996 event. Based on analysis of upstream loading of Zn and Cu found below associated detection levels for the two mainstem sites, raised detection levels (x10) appeared to be just above anticipated levels based on mass balance. No adjustments were made to any data below detection limits. Added bias may therefore be retained in Cu and Zn data at these sites. Final bias accepted would be expected to weaken conclusions linking AUML>2% and metal toxicity, and therefore were interpreted from this more conservative position. Similarly, bias in metal detection limits would have been expected to strengthen conclusions linking active mining and ASML>3% to elevated metal concentrations and toxicity, conclusions ultimately rejected

6.4.3 AMD Water Quality Assessment

To evaluate AMD influenced water quality in Leading Creek, AMD metals were further assessed through water quality assessment. Figures 6.2 through 6.7 present key results associated with Fe, Mn, and Zn in the water column, along with their interactions with TSS, sulfate, and pH. The analysis identifies key signatures in water quality data that relate instream concentrations of various AMD pollutants to land use/cover. To assist in data interpretation, the monthly data collected across Leading Creek and reference sites was segregated by flow regime (Q) and land use. Data was marked by criteria indicating AUML<2%, 2%<AUML<10%, and AUML>10%, where 10% was set as a midway point separating sites TS15 and LCS10 from more severely impacted sites TS16 and TS17. Log-log plots were used to assist in differentiating point data for flow and AUML classifications. Linear regressions accompany some graphics.

6.4.3.1 Zn, Mn, and Drought

Seen in Figure 6.2a and 6.2b, high levels of total Zn was distinguished as the parameter of most concern at AMD sites. Total Mn was also typically a good predictor of AUML and AMD, but not during drought. Mn concentrations in small subsheds increased during drought conditions due likely to increased water evaporation rates and reduced transport rates downstream. Mn is characterized by slow oxidation rates at pH < 9 (Diz, 1997; Cornwell, 1990). For non-drought conditions, Mn and Zn were strongly correlated. Data in Leading Creek and reference sites showed that increasing AUML and decreasing flowrates significantly elevated Zn and Mn concentrations in the water column. Shown in Figure 6.2b, for AUML <2%, elevated Mn trended with decreasing flowrate, comparable during drought in smaller subsheds to the levels seen at severely AMD impacted sites during low to medium flow conditions.

Drought is a spatially dependent, loosely defined term representing 40% of baseflow. It was defined to indicate summer-fall patterns observed during this study when small subsheds <10 km^2 and without inundating sediments begin to exhibit dry stream conditions. The drought trend was not readily observed in Zn data where levels instead tended to increase with higher flows at sites with AUML<2%. Low flow at AUML>10% may be exhibit minor Mn increases attributable to drought, though Zn/Mn proportions remained relatively stable for heavily influenced AMD data. TS15 with AUML = 2.8% appeared to exhibit crossover between drought and AMD influence. In Figure 6.2b the lowest flowrate shown of $5x10^{-5}$ $m^3/s/km^2$ was assumed for station TS8B for that data point where actual flowrate was below practical quantitation.

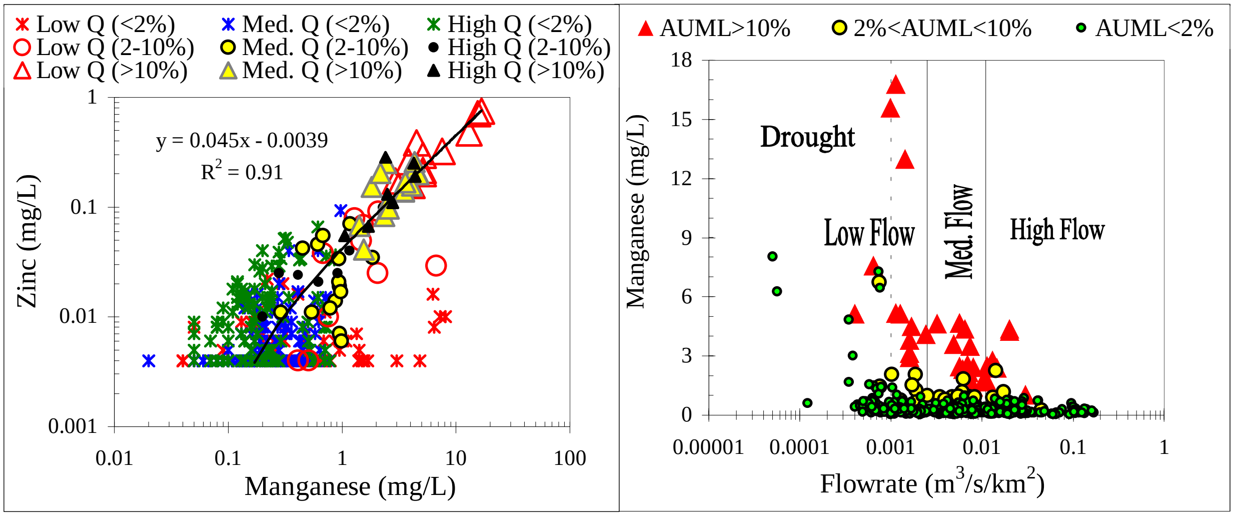

Figure 6.2a and 6.2b – (a) Total Zinc and (b) Flowrate versus Total Manganese – 1996 to 1997 Leading Creek Water Quality Results; (Data separated based on flowrate and %AUML; curve shown excludes drought data with elevated Mn >1 mg/L)

Figure 6.3a confirmed that increasing trends in Zn and Mn observed also tracked with AMD and decreasing pH. This was indicative of more concentrated AUML coverage and lower flows associated with baseflow discharge. Evident in Figure 6.3b, trends were not readily observed between total Mn in the water column and total suspended solids (i.e. washload particles <0.125 mm in size). An exception for AUML<2% at higher flow was evident where a minor suspended fraction could be seen. Robbins (1997) studied the effect of Mn loading on streams in Wayne County West Virginia. Robbins concluded that: a) the most abundant source of easily soluble Mn was coarse-grained sandstone particles in close proximity to coal, b) total Mn instream increased by an order of magnitude during high flow compared to low flow, and c) total Mn was twice as high as dissolved Mn in statistical analysis of county stream data taken from the USGS-WRD WQDATA database.

Robbins indicated bed sediments, particularly rocks with black and brown oxide coatings were the suspected source of Mn flux into the water column during the higher flow events for their samples. The distinction between results here and Robbin's study may be tied to coarser suspended bed loads (e.g. 0.125 mm < fines < 2 mm). Leading Creek washload results were therefore not necessarily inconsistent with Robbins study. Specific grab sampling technique in the USGS data examined was not discussed and disparity was reported among sample sizes, indicating potential bias by unmatched data. In the USGS database, maximum total Mn was 1.9 mg/L and maximum dissolved Mn was 2.0 mg/L. Average total Mn was 0.13 mg/L (n=187) and average dissolved Mn was 0.06 mg/L (n=268). Wayne County average levels represented the lowest tier of data shown in Figure 6.3b. Comparing upstream data, Robbins concluded biodiversity at the downstream site was not significantly impacted by AMD. Thus it would be assumed that the study site was markedly different from Leading Creek sites with AUML>2%.

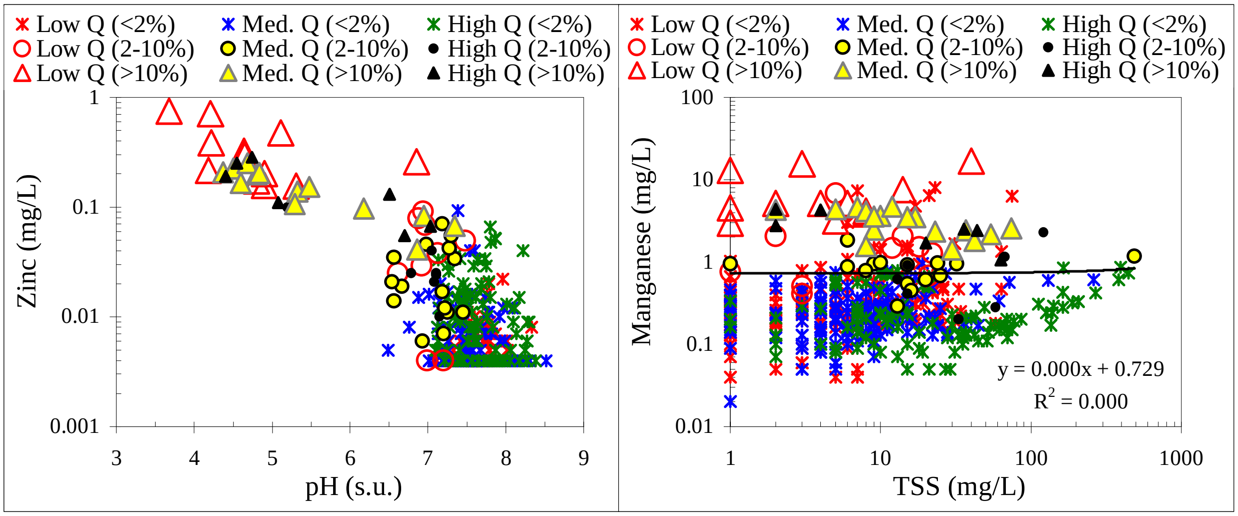

Figure 6.3a and 6.3b – (a) Total Zn versus pH and (b) Total Mn versus TSS – 1996 to 1997 Leading Creek Water Quality Results; (Data was separated based on flowrate and %AUML)

6.4.3.2 Fe, TSS, and Run-Off

Comparing results for TSS to Fe in Figures 6.4a and 6.4b, a definable trend relating elevated Fe concentrations in the water column to elevated TSS during high flows was observed at sites without severe AMD, with exceptions noted at TS17 and other AMD impacted sites. A separate evaluation of STORET data for southeastern Ohio watersheds for the period between 1990 and 1998 also showed this trend. Like TS17, some sites with pH below 6 to 6.5 exhibited above average Fe/TSS ratios. Data in Figure 6.4b is from the same Ohio watersheds evaluated for sediment in Chapter 5 included sites within the Hocking River Basin and southeast tributaries of the Ohio River. In Leading Creek, station TS17, unlike TS16, was subject to continuous precipitation of red-colored iron oxyhydroxide flocs layered across many areas of the streambed during the study. Data in Figure 6.4a appears in part to be reflecting a relative increase in iron content associated with these portions of washload easily entrained during run-off conditions. Fe(III) is suspected as the dominant species since the higher flow data imparts $[H^+]$ dilution. This typically resulted in higher pHs of 5 to 7 s.u., indicative of abiotic oxidizing conditions.

Initial iron precipitate oxyhydroxide flocs, as opposed to final iron oxide coatings on sand particles such as hematite, were best described in Leading Creek as low-shear with respect to natural silt/clay size particles in similar size ranges. This latter conclusion was based on visual evidence gathered over multiple field studies conducted instream. In addition to re-suspended iron flocs, increased Fe(II) and to a much lessor degree soluble Fe(III) concentrations may contribute to overall increased relative proportions of Fe in relation to TSS seen at TS17. Based on solubility of Fe(III) at pH 4.5 to 5.5 during lower flows, assuming dominant species as $FeOH^{2+}$, less than 0.1mg/L would be expected as soluble Fe(III) (Benefield *et al*, 1990). In contrast all or part of soluble Fe in Figure 6.4a may be attributable to soluble Fe(II) species at those pH ranges (Snoeyink *et al*, 1980). Because dissolved iron data was not available, TS17's excess iron above the nominal trendline for TSS could not be directly distinguished from Fe(II). The trend with TSS though for medium and higher flows under abiotic oxidizing conditions implicated solid Fe(III) (Snoeyink *et al*, 1980).

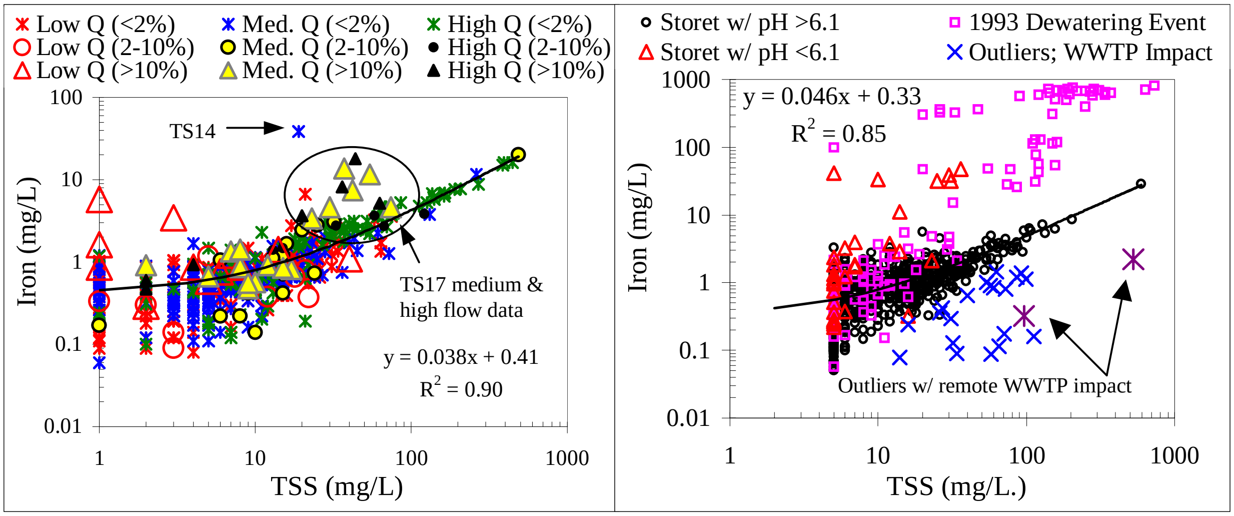

Figure 6.4a and 6.4b – Total Iron versus TSS for (a) 1996 to 1997 Leading Creek Water Quality Results and (b) Southeast Ohio Watersheds; (Data was separated based on flowrate and %AUML; curves shown excluded (a) all medium and high flow data at TS17 and the outlier at TS14 collected on 8/22/96 after heavy storms, and (b) WWTP outliers)

Figure 6.4b includes STORET data available from 1990 to 1998. Outlier data with nearby upstream wastewater treatment plant (WWTP) outfall influences (and other WWTP data included in curve fitting) may indicate plant upsets or the presence of refractory particulate organic waste resulting from high levels of biological wastewater treatment. The culled WWTPs represented a combination of sanitary, industrial, and combined sewer overflows. One industrial facility, a pizza food processing center, was responsible for 67% of WWTP data omitted. Remote downstream data may also be impacted by persistent organic particulate or may simply represent anomalous data. Data shown in Figure 6.4b for the Mine No. 31 AMD dewatering event included some data before and after the 6-week AMD dewatering discharge period. Results indicated that extremely elevated levels of iron were discharged from Mine Mo. 31 instream where maximum total Fe reached 811 mg/L at a pH of 6.5. Values of pH ranged as low as 2.86 s.u., with many samples reported in the range of pH 4 to 6 s.u.. The 10 highest iron values trended with higher pH (average 5.73; σ = 1.0) indicating abiotic oxidizing conditions. Higher iron followed increases in TSS, indicating significant portions were solid Fe(III). Sediment data and circumstances of the Mine No. 31 dewatering were discussed in Chapter 5.

6.4.3.3 Zn/Fe, Mn/Fe, and Sedimentation Mechanisms

Shown in Figure 6.5a, the constant slope relation derived in Chapter 5 for LogZn/LogFe in sediment was compared to Zn/Fe water column data, distinguishing medium and high flow conditions, along with trends for higher zinc found at lower flows. Sediment curves from Figure 5.2a and 5.2b in Chapter 5 are presented in Figures 6.5a and 6.5b, respectively. Original curves and a companion curve assuming intercept = 0 were shown for comparison assuming a background aspect of sedimentation might be involved. In the case of iron, the intercept was positive, and for manganese it was negative. The sedimentation analogy drawn was based on an assumption that mass and molar ratios should each be similar in each phase, less the presence of Fe(II). Figure 6.5a confirmed consistency in water quality data with the sediment relationship found between Zn and Fe in Figure 5.2a. Recall Figure 6.4a indicated high Fe at non-AMD impacted sites associated with run-off conditions was attributed largely to suspended silts/clays.

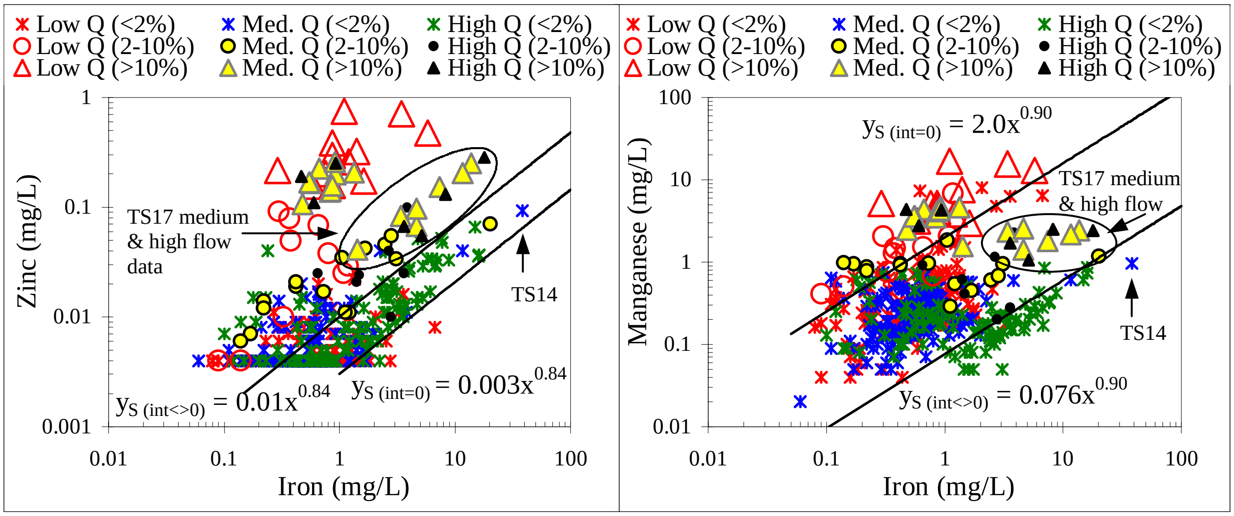

Figure 6.5a and 6.5b – (a) Total Zinc and (b) Total Manganese versus Total Iron – 1996 to 1997 Leading Creek Water Quality Results; (Data was separated based on flowrate and %AUML; curves represent sediment relationships with and without intercepts)

Figure 6.5a indicated the degree to which soluble Zn and Fe(II) are related under more severe AMD conditions and lower flows. An underlying dilution gradient with increasing pH and transition between solid and dissolved phases induced by changes in flowrate is implicit. TS17 appeared to represent combined aspects of soluble Fe(II) but mostly suspended Fe(III) during medium and higher flows. These would be expected to originate from iron-zinc rich flocs entrained into and precipitated from the water column. Water quality concentrations of Fe, Zn, and to a lessor degree Mn changed with land use when approaching abiotic oxidizing conditions, but ratios with respect to iron did not, supporting results in Figure 5.2a and 5.2b in Chapter 5. Abiotic iron oxidation and precipitation rates would be expected to increase sharply during run-off events as pH is raised from 4 to 6 to more neutral levels, further sinking zinc to solid phases.

Zn is related to sedimentation processes associated with organic materials, iron, and clay mineralogy (Benefield *et al*, 1990). This was observed in natural settings supported by findings for sediment in Figure 5.2a in Chapter 5 where consistent LogZn/LogFe ratios were observed. Mn precipitation would be expected to be associated with extremely slow microbial oxidation processes and oxide coatings on larger sediment particle surfaces (Diz, 1997; Robbins *et al*, 1997; Cornwell, 1990). An immediate question that arises is what are the sources of elevated Mn found in the water column when depressed Mn was sometimes found in bed sediments with severe AMD? Based on Figures 5.2b in Chapter 5 and Figure 6.2a, and slow biotic and abiotic Mn oxidation kinetics *in situ*, data indicates a greater likelihood that Mn in sediment is dominated by soil origin, not instream precipitation. Paucity of Mn in sediment at severely impacted AMD sites was most likely due to instream dissolution, with high levels of soluble Mn *in situ* also possibly supplied from sandstones on the landscape coming into contact with AMD.

Figure 6.5b more clearly highlights the suspected relationship between insoluble oxidized manganese, iron and TSS during medium and high flow run-off conditions at non-AMD sites. Disparity in the zero-intercept curve for manganese and iron in Figure 6.5b further confirms that different mechanisms of sedimentation are associated with zinc and manganese in the instream environment. Results for the outlier at TS14 (with AUML=1.7%) indicated that the outlier data point in Figure 6.4a was not likely indicative of AMD run-off. The data point was associated with neutral pH and extremely elevated phosphorous and copper indicative of agricultural loading, where error in the TSS value obtained was suspected. This analysis points to an inherent potential benefit in improved quality assurance by examining data from AMD landscapes in this manner. Though inconclusive, evidence built to a more reliable interpretation.

6.4.3.4 Interaction of Fe, Zn, Mn, Mg, SO_4

Figures 6.6a and 6.6b show the product factor [Zn•Mn]0.5 and Zn/Fe ratio and their ability to better differentiate AMD across varying degrees of AUML when complicating aspects such as drought and iron-zinc rich TSS are present. Figure 6.7a and 6.7b also illuminate the role of sulfate in differentiating among various mining influences.

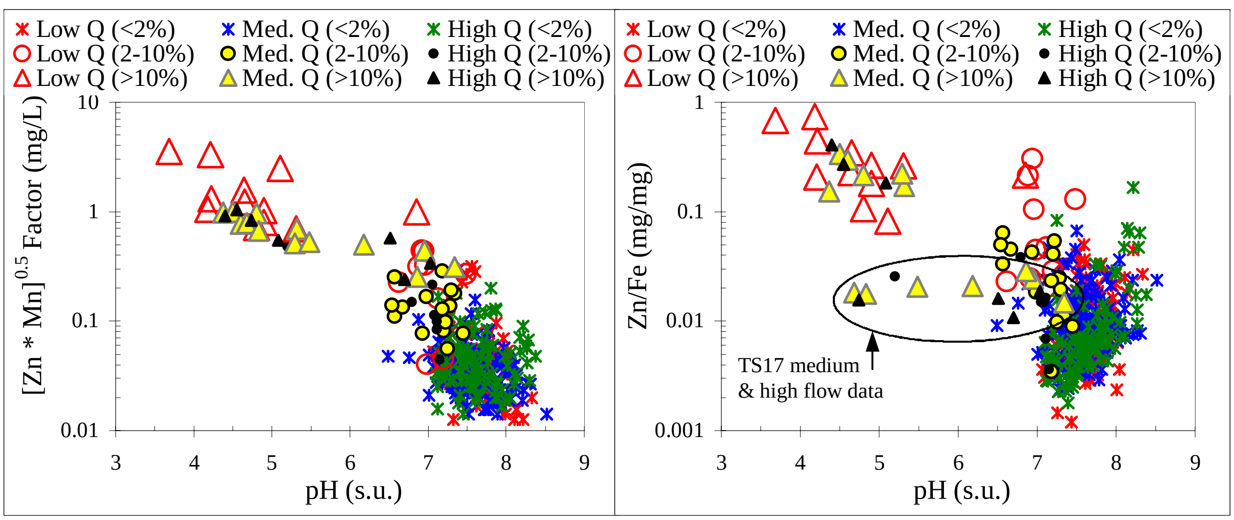

Figure 6.6a and 6.6b – (a) Zn•Mn and (b) Zn/Fe versus pH – 1996 to 1997 Leading Creek Water Quality Results; (Total data was separated based on flowrate and %AUML)

While pH, total Zn and total Mn appeared to be quite useful in distinguishing AMD, metal toxicity and bioavailability will be more accurately assessed within a framework that accounts for solids. Product and ratio constructions may also allow more informed analysis in the presence of non-AMD loading of metals or metal and hydrogen acidity. Of interest here was also the relationship found between magnesium and sulfate and its ability to uniquely distinguish between untreated AMD, treated AMD, and limestone quarry operations at the headwaters in the Leading Creek. This is described in Figure 6.7a. AMD behavior *in situ* is summarized in Figure 6.7b relating dilution and suspension trends in Zn, SO4, flowrate, and land use. The AMD trend in Figure 6.7b implies soluble components. For interpretation of flow segregated data, station TS11 was almost always represented by high flow conditions for site-specific determinations. Figure 6.7a is exploded in four views in Figure I1 in Appendix I looking at each mining signature and background data set individually. Figure 6.7a presents all data where signature functions shown represent fits of individual data sets shown in Figure I1 in Appendix I.

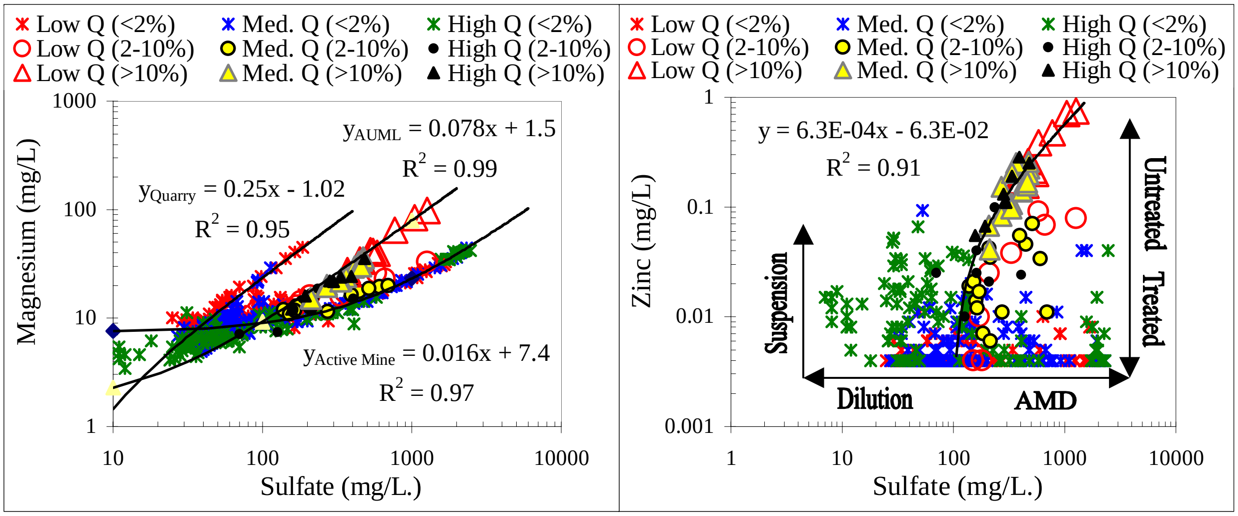

Figure 6.7a and 6.7b – (a) Total Mg and (b) Total Zn versus Sulfate – 1996 to 1997 Leading Creek Water Quality Results; (Data was separated based on flowrate and %AUML; Zn = f(SO_4) curve was for AUML>0 and excluded sites with active mine influence)

6.5 DISCUSSION

Examining soluble phase relations, elevated Zn and Mn tracked lower pHs and represented strong indicators of AMD impact except during drought conditions. Soluble Mn can increase dramatically at sites with no AMD present during drier conditions. The occurrence of severest AMD with baseflow supported the conclusion that AUML is the dominant source of AMD in Leading Creek and not ASML. A major point illuminated here was that Zn, Fe and TSS in the water column were again found to be interrelated for conditions favoring solid phase co-precipitation or entrainment from solids scoured from fine sediment. Thus analysis of water quality data supported results in Chapter 5 where Zn tracked Fe in fine silt/clay size sediments comprising washload monitored as TSS.

Also supportive of results found in previous study of coal bed and sediment data, Mn was again not consistently attributable to Fe and TSS sedimentation dynamics. Correlation of Mn with TSS were observed only at low levels and only under conditions were coarser sand fractions might be entrained during more turbulent higher flows. Ubiquitous sandstone was expected to be the primary source of Mn in Leading Creek. Results here supported a conclusion that sandstones represented the source of soluble Mn instream, where instream acidification and dissolution was the most likely cause of Mn paucity in sediments at some sites with stream acidification where pH <6. Based on analysis of water quality, precipitation of Mn was not observable where biotic and abiotic kinetics would not be expected to favor Mn removal from the water column.

6.5.1 Fe, Zn, and Oxidation-Reduction Processes Instream

Station TS17 on Thomas Fork with average pH 5.5 was visually the most heavily influenced by yellow-orange-red colored oxyhydroxide precipitates, and a consistent ubiquitous presence of hematite coated sands (Nordstrom, 1982). Station TS16 with minimum pH 3.7 and average pH 4.6 tended to exhibit gray precipitates where yellow and red precipitates were never observed. An explanation of sediment concentrations for TS16 and TS17 and other sites may be found in comparably slower kinetic rates associated with autocatalytic oxidation of Mn(II) than for Fe(II) at pH 4 to 7 (Cornwell, 1990). The scenario is further governed by increasing oxidation rates for Fe(II) with increasing pH in this range. Cornwell showed that abiotic oxidation of Mn is negligible below pH 9, where instream oxidation is largely determined by microbial processes (Robbins *et al*, 1997). Relatively less abiotic oxidation of Fe(II) at pH 4.6 will occur with shorter residence times for baseflow that prevail in 1st order tributary reaches like TS16 compared to 4th order subsheds monitored with higher pH like TS17. At TS17 with average pH 5.5, though still slow kinetically, appreciably more abiotic oxidation of Fe(II) and subsequent precipitation of Fe(III) will occur (Nordstrom, 1982; Williamson *et al*, 1994). Similarly, more soluble Fe(III) will precipitate at average pH >5 (Snoeyink *et al*, 1980).

Benefield (1990) indicated co-precipitation of iron-metal complexes involving Zn, organic materials, and clay mineralogy is significant where solubility of Zn is dramatically reduced in the presence of Fe(III). Abiotic Fe(II) oxidation and Fe(III) precipitation rates would also be expected to increase dramatically during run-off events as pH is raised from 4 to 6 and higher, further sinking zinc to solid phases (Diz, 1997; Snoeyink *et al*, 1980; Williamson *et al*, 1994). Expectations were supported in natural stream settings with AMD as seen in sediment in Figure 5.2a in Chapter 5 where consistent LogZn/LogFe ratios were observed. In Figure 6.5a, accounting for a linear model intercept offset, at higher flows, water column Zn and Fe tended to maintain similar proportions in non-AMD environments. This was also true for AMD environments under conditions where solid forms were characterized through TSS data

It was previously shown in Leading Creek that Fe oxidation and precipitation maintained similar LogZn/LogFe ratios in sediment across variable drainage areas, land uses and extreme levels of stream acidification, with only minor variation noted for ASML>3%. Fe sinks to sediment as oxyhydroxide flocs and will co-precipitate and adsorb sulfate, enhancing delivery of sulfate to the streambed (Mills *et al*, 1989). Results correlating Fe, Zn, and TSS therefore are reasonably expected phenomena in AMD and non-AMD environments. Mills also showed how Fe(III) and sulfate could lead to stable de-acidification of the sediment column through reductive processes, essentially reversing pyrite oxidation. Thus sediment columns within severely impacted AMD reaches tend in equilibrium to reach neutral pH ranges just below the water-sediment interface and can stabilize Fe in solid forms.

6.5.2 Origins of Fe, Zn, and Mn

Coal bed, sediment and water quality data in Leading Creek indicated that elevated sources of soluble Zn and Fe found at sites with AUML >2% can easily originate from clay/shales in close proximity to coal bed seams. Mn precipitation would be expected to be associated with extremely slow microbial oxidation processes and oxide coatings on larger sediment particle surfaces (Robbins *et al*, 1997). An immediate question that arises is what are the sources of elevated Mn found in the water column when depressed Mn is sometimes found in bed sediments? Figure 5.2b in Chapter 5 and knowledge of relatively slow biotic and abiotic Mn oxidation kinetics *in situ* indicated a greater likelihood that Mn in sediment is dominated by soil origin. Thus with respect to Fe, it was found here that instream Mn loading to sediment from high Mn in the water column was minimal. This was further evidenced even at higher pH at non-AMD sites where drought data showed elevated Mn. In the natural setting, paucity of Mn in sediment at severely impacted AMD sites would be more likely attributable to dissolution, with elevated soluble Mn likely also supplied from sandstones on the landscape (Robbins *et al*, 1997).

Water quality and sediment data suggested that some portion Mn in the water column was dissolved from instream sediments at sites with AUML>10%. Based on Robbin's work (1997), Mn would be expected to originate from ubiquitous sandstones found originally as cap rock in Leading Creek. Discussed in Chapter 3, eroded sandstones currently inundate AUML and ASML impacted streambeds due to natural erosion and ASML sedimentation. Sandstones were not in close proximity to AUML coal beds, laying 1 m above the zone of oxidation with an interbedded layer of shale. AMD discharging as baseflow may predominantly strip Mn after discharge to streams, or beforehand from sandstones found on the landscape below bench elevation. In the latter case, ample sandstone materials deposited by ASML activities were typically disposed directly below bed bench elevations and could easily intercept AMD seepage at hillside slope toes where AMD is expected to first manifest at the surface. The mechanisms explain how Mn in stream sediments may be unusually low in the presence of severe AMD.

6.5.3 AMD Metals and AUML

Under conditions of elevated Fe during run-off when pHs increase, water quality data indicated that Zn was binding in sediment with iron. Mn on the other hand appeared to experience only dilution during run-off periods. At non-AMD sites during periods of high flow, modest increases in levels of Mn in the water column were likely derived from previously oxidized portions found in soil entrained from the landscape or in sediment. This was consistent with an assertion that oxidized Mn was bound in courser sands compared to Zn and Fe bound in silt/clay sized particles. The implicit assumption is that Mn precipitation instream is minimal with respect to sources of soluble Mn available in the water column due to slow kinetics of oxidation. Consistent with Figures 6.3b and 6.5b, stream mechanics dictates that comparatively less solid Mn would be entrained under low and medium flow conditions, and typically little would be sampled in stream surface washload except under higher, turbulent flow conditions (Vanoni, 1977; Yang, 1996). Thus, most total Mn in stream sampling would be expected to be soluble for all but higher flow conditions.

Including analysis in Chapter 5, it was shown that Zn and Fe in sediment and water column samples can be related in various ways to mineralogy of coal bed ash and clay/silt sized materials in sediment. Strong relations between Fe and TSS in Leading Creek and reference watersheds were also realized in non-AMD subsheds, where urban environments with higher silt/clay content exhibited higher magnitudes of Zn and Fe in similar proportion. It is believed that consistent Zn/Fe relationships seen in AMD and non-AMD subsheds are simply reflecting precipitation dynamics of Zn and Fe(III), which are known to be closely related (Benefield *et al*, 1990). This was also supported by water column departures observed between TSS and Fe for

AMD sites indicating the additional presence of iron-zinc rich precipitates. This latter aspect reconciled with higher Zn/Fe ratios in the water column that were observable only at AMD sites, and total Zn/Fe ratios in the water column that were not directly correlated with Zn/Fe in the sediment column.

6.5.4 Soluble Zn, Mg, and Sulfate

Relationships between magnesium and sulfate within Leading Creek were strongly segregated among limestone and various coal mining land uses excluding strip mining. This was further evidence of the far less significant relationships found between ASML, AMD, and biodiversity in Leading Creek. Figure 6.7b summarizes important aspects of AMD and instream mechanisms. AMD was observable as increased sulfate production above the AUML line given. Zn and sulfate also strongly identified relative AMD stress, where mixing of treated and untreated AMD was observed at mainstem LCS10. LCS10 was subject to both AMD input from Thomas Fork (TS17) and active mine No. 31 flows discharged to the mainstem. Quarry operations were characterized as alkaline drainage with net sulfate increases on the mainstem measurable up to approximately 100 mg/L. Analysis in Appendix C was also presented which showed that the quarry significantly contributed flow to the headwaters during low flow regimes, increasing pH and alkalinity. With higher flows, dilution and suspension concepts were notable and emphasized a gradient in reduced Zn and sulfate with eventual entrainment of fine material bed sediments where increasing Zn was bound with Fe.

6.6 CONCLUSIONS

It was shown that unique instream water column signatures for quarry operations, active underground mining and AML land uses, respectively, could be developed from a knowledge of water quality concentration data, accompanying flow data, and drainage area. Drainage area was used in this analysis as a key variable in flowrate and flow regime estimation, and for normalization of land use activity to surface area. Looking at active and abandoned coal mining, untreated AMD from AUML>10%, and to a lessor degree AUML>2% dominated water quality concerns associated with low pH and elevated Mn and Zn. Unit-area AUML disturbance provided a reliable gradient for quantifying these impacts of AMD instream. The importance and utility of the concurrent examination of quantified flow regime and unit-area land use activity at scales of 1:24000 was demonstrated. Two dominant environmental factors, flow regime and land use, were found to be strong discriminants relating landscape hydrology to trends in water and sediment quality. For a given level of land use, predictable, quantified gradients in flow regime imparted elevated AMD derived soluble metal concentrations during increasingly dryer conditions associated with baseflow discharges.

Baseflow discharges represented the greatest flow regime of concern for AMD in Leading Creek preferentially relating AUML to AMD and not ASML. Identifying characteristics of AMD showed that under most low and medium flow conditions, AMD and low pH led to elevated Zn and Mn instream. High levels of dissolved Mn can be good predictors of AMD, but not during drought. Mn concentrations in small subsheds may increase during drought due to increased water evaporation rates, reduced transport rates downstream, and slow oxidation rates at pH < 9. Total Zn and Mn tended to exhibit levels well above average soluble stream abundance data, where unusually high total Fe was tied to sedimentation. For low and medium flows, total Zn and Mn were determined to be representative of soluble species, where suspended Zn and Fe rich flocs associated with fine materials <0.125 mm can increase in proportion during higher flow conditions. Total Zn data can modestly bias assessment of bioavailabilty during run-off conditions due to entrainment of silts/clays. The conclusion reached was that total Zn/Fe ratios in sediment were non-differentiable with respect to coal mining, and total soluble Zn/Fe may be able to be used to identify soluble Zn without the task of filtering samples. Total Mn was primarily of soluble form during low and medium flows, and increased modestly in suspended form during high flow washload monitoring. Suspended Mn in washload data would be expected to be associated with suspensions of coarser sand fractions in sediment. Based on analysis of water column mechanisms, paucity of Mn in sediments at some sites with severe AMD was attributed to *in situ* dissolution of sandstone materials in the sediment bed.

Magnesium and sulfate signatures within Leading Creek were strongly segregated among limestone quarry and coal mining land uses excluding strip mining. Unique linear relationships were found for limestone quarry operations without AMD impact, for treated AMD without untreated AMD impact, and for untreated AMD without treated AMD impact. Zn and sulfate were also strongly correlated for untreated AMD associated with AUML.

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Chapter 7 - SEDIMENT TOXICITY ASSESSMENT OF URBAN, AGRICULTURE, AND COAL MINING LAND USES USING 10-DAY STATIC WHOLE SEDIMENT TESTS WITH *CHIRONOMUS TENTANS* AND *DAPHNIA MAGNA*

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(ABSTRACT)

Examination of acid mine drainage (AMD) related to *in situ* sediment toxicity was conducted as part of a larger hydrologic-based ecological risk assessment of Leading Creek, located in southeast Ohio. Abandoned mining (AML) involved both underground near-surface mining (AUML) and strip mines (ASML). Active deep long-wall mining operations are still conducted in the watershed today and discharge treated pH-neutralized AMD. Sediment concentration levels were previously found to fall within normal ranges of crustal abundance for %C, Fe, Mn, Cu, and Zn, including sites moderately and severely impacted by AMD distinguished by AUML>2% and 10%, respectively. Standard in-lab whole sediment toxicity test evaluations using *Chironomus tentans* and *Daphnia magna* were investigated looking at survivorship, growth, and neonate production. Impacts upon aquatic biodiversity attributed to urbanization and untreated AMD were shown here to be unrelated to sediment toxicity. Associated with active mining, the watershed's highest sediment metal concentrations were also not an apparent significant source of stress to aquatic biodiversity nor were elevated levels of metals found in sediments at urbanized sites and sites with AMD. Response for *C. tentans* may be modestly biased (+) by AML impacts and sedimentation, while that for *D. magna* may be modestly biased (-) by AUML impacts associated with AMD.

Chapter 7 - SEDIMENT TOXICITY ASSESSMENT OF URBAN, AGRICULTURE, AND COAL MINING LAND USES USING 10-DAY STATIC WHOLE SEDIMENT TESTS WITH *CHIRONOMUS TENTANS* AND *DAPHNIA MAGNA*

7.1 INTRODUCTION

Examining the role of acid mine drainage (AMD) induced toxicity in sediments and the water column within a hydrologic-hydraulic context will lead to more appropriate ecological risk assessment frameworks. This widespread source of pollution today plagues many watersheds in the United States where billions of dollars will be spent in the next 10 to 20 years in an attempt to ameliorate associated consequences of ecological degradation. Both untreated non-point sources of AMD and well-treated point sources of AMD can conceivably cause significant toxicity. Technically sound assessment frameworks are therefore needed to develop successful management strategies for complex ecosystems that define watersheds impacted by AMD.

7.1.1 Risk Assessment of Current and Historic Coal Mining Activities

A multimedia, multidisciplinary hydrologic-based ecological risk assessment framework was utilized in the study of the Leading Creek Watershed. A 388 km^2 stream system located in southeastern Ohio, Leading Creek has been heavily impacted by various coal mining activities dating back to the early 19^{th} century. The stream system provides a unique study environment since 5.5 to 6 million tons of coal are still annually produced today from two active mines that underlie the watershed. An example of the system's unique nature and endemic assimilation of stress, a billion gallons of untreated AMD was released from one of the active mines during an emergency dewatering event in 1993 (Birge *et al*, 1995). An accidental release of silt-sized coal slurry from the active mines' treatment system's polishing pond also occurred in 1997.

To place concerns from active mining in perspective, the Leading Creek Watershed appears to be more heavily influenced by the legacies of abandoned strip mining (ASML) and abandoned underground coal mining (AUML). Like many Appalachian watersheds, coal mining is often commingled with some urban and agricultural activities. This defines a challenging list of possibilities that must be addressed in conducting an informed watershed-scale ecological risk assessment. Prior to this work, questions remained as to what were the dominant stressors in Leading Creek with respect to responses exhibited in aquatic biodiversity. Equally important was determining what the dominant toxicants in this system were, what were the sources and fate of those toxicants, and what were the primary routes of exposure to aquatic biota

Quantitative assessment of land use/cover and its relation to ecotoxicity is not often well defined in many ecological risk assessment programs. Ecotoxicity was studied here for its capacity to relate biodiversity to various coal mining land uses, and to better assimilate the role of water column and sediment columns in risk assessment frameworks that attempt to address AMD. Complimenting previous work done in landscape-habitat and landscape-biodiversity analysis, landscape-toxicity and chemistry-toxicity relationships were further examined. Relationships between sediment chemistry and sediment toxicity in severely impacted AMD reaches were investigated. This work quantified the relatively insignificant role of sediment toxicity *in situ* where severe AMD problems occur and biodiversity is decimated. Following this analysis, a second paper seeks to improve our understanding of related water quality driven ecotoxicological responses to urban, agriculture, active coal mining, AUML and ASML.

7.1.2 Sediment Toxicity and Benthic Macroinvertebrate

Bioavailable toxicants in sediment are expected to significantly impact metrics of aquatic biodiversity. Macroinvertebrate surveys provide an indication of aquatic life living in the sediment as determined by field sampling procedures (OEPA, 1989; USEPA, 1989). Chemical contamination is also most often found in fine "muddy" sediments, which characterize depositional areas (Powers *et al*, 1992). Areas of preferential deposition of potentially toxic sediments in natural alluvial streams include pools, transition zones from pool areas, and point, tributary, middle, and alternating bars which exhibit lower energy and turbulence (Yang, 1996). Riffle-bars of gravelly streams are the most diverse, productive zones within aquatic stream habitat (Allan, 1995; Burton, 1992; Milhous, 1982; OEPA, 1989; USEPA, 1989). Gravel beds are high production zones of biota, and represent critical depositional areas by acting as a trap for fine materials <0.125 mm. Fines infiltrate the gravel framework to certain depths before sealing and backfilling towards the water column – sediment bed interface (Diplas, 1994; Lisle, 1989).

Benthic macroinvertebrates are good indicators of toxin impact to the overall ecosystem, and are increasingly used to assess sediment toxicity (Ingersoll *et al*, 1990; Rosenberg *et al*, 1993; USEPA, 1996). Exposing borrowing organisms to contaminated sediments is thought to represent the worst case real world condition, though debate continues over the most appropriate uptake path to monitor (i.e. pore water, sediment, or water column/pore water interfaces) (Powers *et al*, 1992). These instream zones are intimately related by equilibrium chemistry and stream mechanics. Temporal and subshed-scale spatial variation is exhibited in toxic responses to watershed pollutant loading, in some cases originating far upstream (Dunne *et al*, 1978; USEPA, 1996). Holistic aquatic ecosystem risk assessment must at some level discern relative importance of contaminant phase partitioning over space and time (USEPA, 1996, 1998).

7.1.3 Toxicity Testing Study Objectives

To evaluate AMD in an expanded risk assessment framework encompassing ecotoxicity directly, collection and comparison of spatial and temporal data describing four profiles of the watershed were undertaken and spanned several years of monitoring. Fundamental watershed traits chosen covered (a) hydrology-hydraulics, (b) chemistry of natural sediments and waters, (c) insect and fish community assemblage, and (d) comparison of responses to toxicants shown for lab and *in situ* test organisms. This paper describes the sediment tier of a three-tiered toxicity test program undertaken within the integrated risk assessment. The full experimental design for ecotoxicological assessment included 10-day in-lab whole sediment tests, 48-hour in-lab whole effluent acute toxicity tests, and 35-day chronic *in situ* toxicity tests. The three toxicity tests were used to assess effects of land use/cover upon aquatic biodiversity.

The hypothesis investigated here addressed aspects of hydrologic-based watershed-scale ecological risk assessment by employing Geographical Information System (GIS) land use/cover analysis. The objective was to evaluate quantitative descriptions of urban, agriculture, surface mining, and underground coal-mining activities, and to relate their impacts to ecotoxicity. Of interest overall was defining landscape-toxicity and toxicity-biodiversity dynamics. Both active and abandoned mining activities were considered. Few active and abandoned mine studies exist that combine GIS at fine scales of 1:24000, land use/cover analysis, and ecotoxicological risk assessment tasks. This paper integrated these approaches by examining the hypothesis that impact upon aquatic ecology attributed to AMD from AML, and treated AMD from active underground mining are functions of water quality degradation and not sediment quality. Investigation of the hypothesis was fully integrated by comparing land use/cover to measures of benthic macroinvertebrate biodiversity, and sediment and water column chemistry and toxicity.

7.2 STUDY WATERSHEDS

Leading Creek is a fifth order stream in Ohio (scale: 1:24000) 96% of which spans Meigs County, with other portions in Athens, and Gallia Counties. Together, 11 mainstem stations and 19 tributary stations shown in Figure 7.1 were selected for hydrologic, hydraulic, and ecological risk assessment monitoring. Mainstem stations were located to provide complete longitudinal coverage of Leading Creek, and discrimination among potential sources of sediment or contaminant impacts to the stream system. Monitoring also included 3 reference sites.

7.2.1 Leading Creek Mainstem and Tributary Stations

Ten mainstem sites and six of the tributary sites were included in stream stage monitoring indicated by designation as staff gage stations in Figure 7.1. Agricultural activities dominate the upper mainstem while downstream reaches are dominated by mining. Eighteen tributary stations in Leading Creek received some level of consistent sampling effort for most variables described here. Monitoring stations were initially chosen based on association with either (1) their known or suspected role in the contribution of point source pollutant loading to Leading Creek, or (2) their significance based on the size and nature of associated drainage area. Parker Run (TS11) and Ogden Run (TS6) in Figure 7.1 receive active coal mining treatment plant effluents and associated stormwater and sanitary discharges. Other NPDES sites relate to relatively insignificant sanitary treatment (STP), water treatment (WTP), and quarry discharges. A closed landfill is located along the mainstem just above the reach where the quarry discharges.

7.2.2 Ecoregion Reference Watersheds

Three comparative reference stream sites in two watersheds were selected on the basis of (1) exceptionally high IBI/ICI biodiversity scores in historic assessments, or (2) visibly exceptional habitat characteristics that are representative of what might be attainable in the Western Allegheny Plateau ecoregion. IBI refers to the site’s Index of Biotic Integrity, a fish-based metric, and ICI refers to the Invertebrate Community Index, an insect-based metric (Karr *et al*, 1991; Miller *et al*, 1988; OEPA, 1989, 1991a, 1991b; USEPA, 1996). Upon initial inspection, the reference sites did not indicate marked decline but were potentially impacted by various anthropogenic traits including effects from urban, agricultural, ASML, or AUML sources. Due to the wide reach of AML in Appalachia, there were no readily identifiable similar sized watersheds nearby that could claim strong independence from all or most of these stresses.

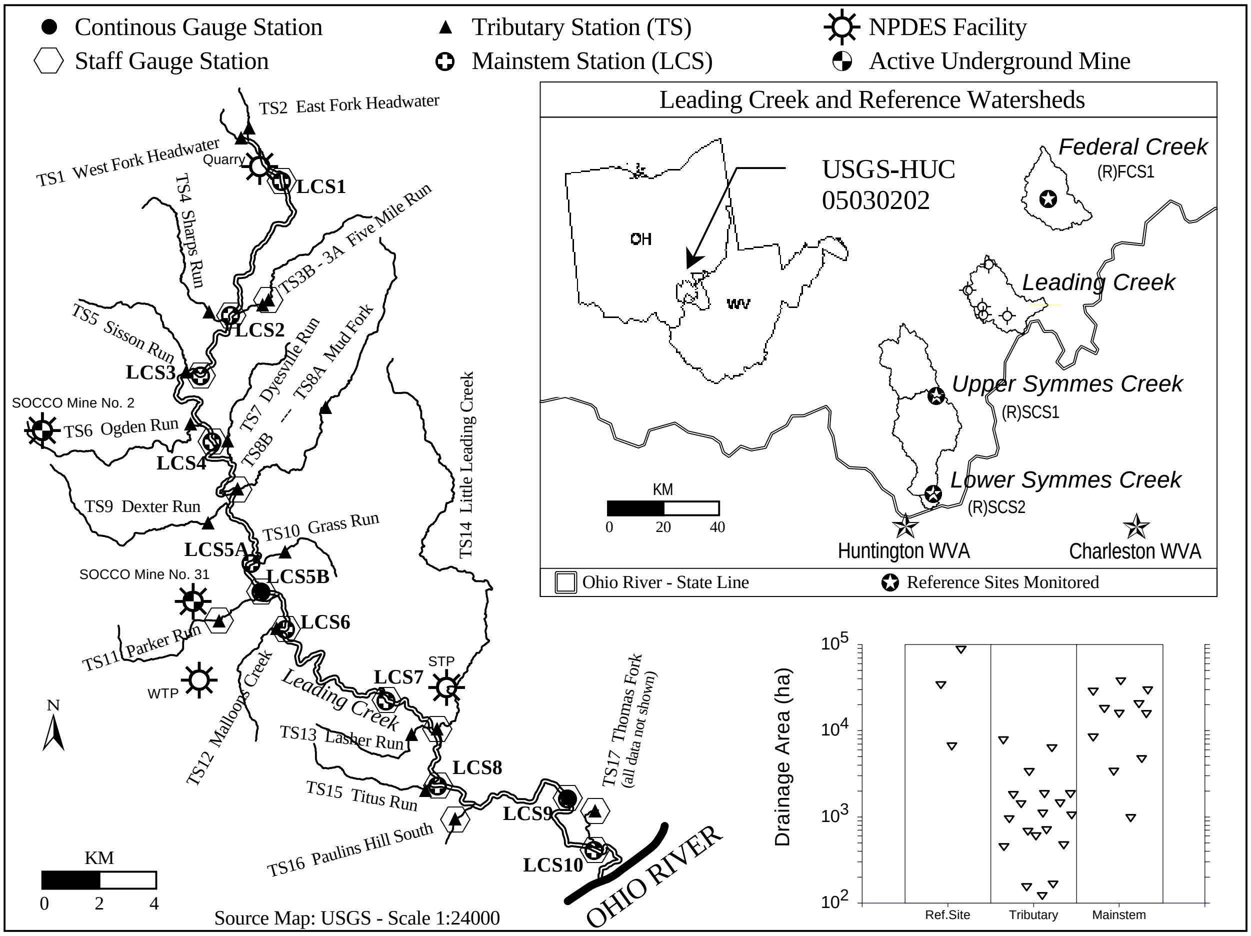

Figure 7.1 - Leading Creek Monitoring Network and Reference Watersheds

7.3 METHODS AND MATERIALS

Data collection in Leading Creek and the reference watersheds included monitoring of: (1) precipitation, stream stage, water velocity, and flow; (2) sediment and water quality; (3) benthic macroinvertebrate and stream habitat; and (4) toxicity through sediment, acute, and *in situ* testing programs. Toxicity testing included chronic in-lab 10-day sediment testing using 2 species: *Chironomus tentans* and *Daphnia magna*, a midge and cladoceran, respectively. To assess chemical, physical, and biological interactions, stations in Figure 7.1 covered a wide range of drainage areas and stream orders (Dunne *et al*, 1978; Minshall *et al*, 1985; USEPA, 1996; Vannote *et al*, 1980). Data collection methods specific to qualitative habitat, biodiversity, sediment quality, sediment toxicology, and land use and land cover are outlined.

7.3.1 Land Use and Land Cover

Land use/cover data (scale 1:24000) was developed through three sources including a) 7.5-minute quadrangle series maps depicting ASML, b) Ohio Division of Natural Resource (ODNR) geodata depicting AUML, and c) June 1988 Landsat5 Thematic Mapper (TM) geodata developed by ODNR specifically for assessment of abandoned mine lands. Because of lack of utility found in previous examinations of landscape-biodiversity relationships, 1994 Landsat5 TM geodata developed by ODNR for general environmental assessments was omitted from this analysis. This work focused on directly sensed portions of geodata characterizing abandoned mined lands due to its proven strong representation of AML as a significant source of stress to aquatic habitat and biodiversity. Remotely sensed TM land use/cover variables considered were Mine Gob (darkest coal wastes), Mine Spoil (coal waste mixed with subsoil and topsoil), Bare Soil (representing heavy agricultural activities), Sparse Vegetation (<50%), Moderate Vegetation (50 to 100%), Full Vegetation (100%, mostly herbaceous), Shrub, Wooded, Water, Wetland, Urban. Directly developed data evaluated included AUML, ASML, and unreclaimed ASML.

7.3.2 Chemistry of Natural Sediments and Water

Sediment sampling was conducted at all mainstem, tributary, and reference sites shown in Figure 7.1 except for station LCS5A used for additional biological data collections only. Fine sediment samples were collected in 1996 and 1997 using standard grab sampling techniques in accordance with standard methods for toxicity testing (APHA, 1992; ASTM, 1990). At each site 3 to 5 replicate samples were collected from a representative depositional area along a cross-sectional transect, restricting sampling to the upper 1 to 10 cm of bed depth (Burton *et al*, 1992). Sediments were collected by wading cross-sections except in 1996 where stations LCS9 and LCS10 required use of an Ekman dredge lowered from bridgedecks. Water and sediment depths (Gallagher *et al*, 1999) were collected during each cross-section sampling event. Except for 13 sites where 1997 sediment chemistry was evaluated from the 1st quarter 1998 samples, all chemistry and toxicity data here were generated from split samples.

Sediment samples were placed in sterile bags and returned on ice to the Environmental Laboratory Systems facility at Virginia Tech where they were refrigerated at <4 ^{o}C until analysis was completed. To manage requirements for two-week holding times for sediment toxicity tests, the watershed was split into upper, middle, and lower sections and sediment was sampled on three different dates each year. Sediment samples in 1996 and in most cases for 1997 were split for sediment toxicity testing and chemical analysis was completed for %carbon, Fe, Mn, Zn, and Cu. Testing of metals in solid samples collected were analyzed using acid extraction followed by analysis with inductively coupled plasma (ICP) (APHA, 1992). Symmes Creek was not evaluated for sediment chemistry where some background data was available (OEPA, 1991b). Previously evaluated, casual factors of poor biodiversity associated with water quality were also analyzed including pH, specific conductivity, selected total metals, and other water quality parameters characterizing urban, agriculture, and coal mining pollution.

7.3.3 Benthic Macroinvertebrates

Various biological sample collections and habitat surveys were carried out during appropriate spring and fall seasons. Supporting habitat sampling followed protocols outlined in the US EPA's Rapid Bioassessment Protocols (RBP) (USEPA, 1989). Habitat was also evaluated using Ohio EPA's Qualitative Habitat Evaluation Index (QHEI) (Rankin, 1989). RBP and QHEI habitat evaluations were supplemented with quantitative instream and riparian surveys conducted in fall of 1996 and spring and fall 1997 (Babendreier *et al*, 1998; Gallagher *et al*, 1999). Riffle, pool, and shoreline habitats at each station were sampled intensely using D-framed nitex-mesh dipnets, and samples preserved in ethanol (OEPA, 1989). Identifications were made with the aid of various taxonomic keys (Mason 1973; Merrit *et al*, 1984; Pennak 1989; Wiggins 1977). Taxonomic values derived included total taxon richness and abundance as well as analysis of Ephemeroptera-Plecoptera-Trichoptera (EPT) abundance and EPT richness. For collection of benthic macroinvertebrates, tributary and mainstem sites were sampled twice in 1996 (EPT 2x); Parker Run and mainstem sites three times in 1997 (EPT 2x); and all sites below Mine 31 twice in 1995 (EPT 0x).

Ecological variable analysis followed previous works of Richards and Host (1994b) by examining landscape-habitat relations for habitat variables significantly correlated to biodiversity using Spearman-rank analysis. A concept of threshold land use/cover (e.g. ASML>3% by area) was employed to differentiate sites via analysis of variance in order to assess significant trends in qualitative habitat data and biodiversity. Both parametric and non-parametric tests were needed to meet assumptions for analysis (Zar, 1999). Landscape-biodiversity interactions were also assessed through correlation analysis and threshold categories were similarly evaluated through ANOVA. Threshold criteria for land use/cover were typically selected for study where aereal extents were first normalized to drainage area. These included AUML=2%, ASML =3%, Urban =5%, Bare Soil =3%, and Boolean functions based on active mining land uses.

7.3.4 Sediment Toxicity Testing

Directly developed and remotely sensed land use/cover geodata were analyzed within a three-tiered toxicological testing program that looked at both stream water and sediments. Sediment toxicity testing was coordinated with sediment chemistry and fine particle size analysis for two events at each site, once each in 1996 and 1997. Toxicity evaluations included chronic in-lab 10-day static non-renewal sediment tests using *C. tentans* and *D. magna*. Protocols used for *C. tentans* and *D. magna* chronic tests were based upon standard sediment testing methods (ASTM 1990; Nebeker *et al*, 1984). Standard practices were used for sediment sub-sampling, preparation and handling, and reference site diluent addition (Babendreier *et al*, 1997).

For each site during the second event, 200 g of sieved sediment (<2 mm) was added to 1 L beakers. Federal Creek reference site diluent was then added to a capacity of 1 L. For the first event 1:4 sediment-diluent ratios were also used but 80 g of sediment was placed in a 600 ml beaker. After a 24-hr stabilization period, 10 midges (10 days old) were added to each beaker except for the initial set of tests that used 5 midges each. After 10 days, *C. tentans* mortality was recorded, and surviving larvae dried and weighed to determine growth. Dissolved oxygen, pH, specific conductivity, and temperature were monitored throughout the test. Dissolved oxygen was maintained at or above 4.0 mg/L by aeration with a pipette placed just below the water surface. Procedures for *D. magna* were the same, except five 5-day old daphnids were used. Parent survivorship and neonate output for *D. magna* were determined.

Additional toxicity replicate data was generated from 3 sample sets for some sites in 1996 and 1997. Direct comparisons between split-sample toxicity and chemistry subsection data were avoided due to missing split samples; uncertainty regarding subsection identification for added replicates; and strong subsection serial correlation. Midge growth was calculated as the average weight of recovered organisms per sample vessel, from which further statistics were developed.

7.3.5 Description of Statistical Tests

All parametric normality tests described here included skewness, kurtosis, and combined omnibus tests (Zar, 1999) available within the NCSS97 software package (Hintze, 1997). Unless otherwise specified, a significance level α = 0.05 was employed. Where used, parametric ANOVA between group means was supplemented using the Tukey-Kramer multiple comparison test with α_{MC} = 0.10. In cases of non-normality of residuals or heterogeneity of variance, Kruskal-Wallis one-way ANOVA ranks were performed testing significant differences between group medians using data corrected for ties. For non-parametric ANOVA, the Kruskal-Wallis multiple comparison z-value test was used to evaluate significant ANOVA results employing the full Bonferroni criteria for all groups (Hintze, 1997, Zar, 1999).

7.4 RESULTS

With average pH 6.7 and 7.3, respectively, TS15 and LCS10 were less impacted by AMD than stations TS16 and TS17 with average pH 4.6 and 5.5, respectively. LCS10 is the only mainstem site with AUML>2%. While station LCS10 pH is well buffered from upstream mainstem alkalinity inflow, significant water quality and sediment quality impacts were consistently observed giving it distinction as an AMD impacted site. Sites with AUML>2% are distinguished for later use in comparing land use/cover to sediment and water ecotoxicity. For all other sites in Figure 7.1 except LCS1 and TS11, average pH under all flow conditions ranged between 7.4 and 7.7 s.u. (σ = 0.2 to 0.4 s.u.). At TS11 below active Mine No. 31, average pH was 8.1. LCS1 pH was 7.9 due to the quarry. The Federal Creek site had average pH 7.8, and both Symmes Creek sites had average pH 7.3. For most time periods, Mine No.2 wastewaters are conveyed to Mine No. 31 and treated process wastewaters discharged to Parker Run (TS11).

7.4.1 Land Use/Cover and Relationships to Habitat and Biodiversity

Summary findings from detailed analysis of landscape-habitat-biodiversity relations for Leading Creek are presented here for background in assessing ecotoxicity data. In general, strong relationships were observed between benthic macroinvertebrate taxon richness and abundance for both total data and EPT groups, respectively. Landscape-habitat relationships are summarized in Table 7.1. Average spring, summer and fall reach results (riffle+run+pool) for qualitative dipnet collections of benthic macroinvertebrates are given in Tables 7.2 and 7.3 with comparisons to various land use/cover categories. Examining significant relations between landscape and biodiversity, Table 7.2 presents results for urban and severe AML land uses. Some comparisons indicated as significant in Table 7.2 were qualified by borderline rejection of a parametric test assumption, or by omission of the group ASML<3% from the population.

Detailing insignificant relations found between landscape and biodiversity, Table 7.3 presents summarized results which focused on assessment of agricultural (AG) activities, minor strip mining activities where ASML<3%, and active mining operations in Leading Creek. In Table 7.3, no significant differences were noted between any groups defining minor strip mining activities or agricultural activities. In these comparisons, both minor ASML and agricultural focuses excluded sites with ASML>3% (and by default AUML>2%) and sites with Urban>5%. These resultant focus groups were both compared against active mining and reference site data.

For the urban and more severe AML focused comparisons shown in Table 7.2, active mining was not distinguished and all sites were compared against reference data.

Table 7.1 –ANOVA Results for Land Use/Cover-Habitat Relationships

Land Use/Cover Category	# Sites	Avg QHEI			Significant Group Comparisons					
		(T)-Total	(S)-Substrate	(P)-Pool/Riffle	Urban >5%	ASML >3%	AUML >2%	Bare Soil >3%	BareSoil <3%	Active Mines
Urban >5%	2	43	1.0	5.0	-	-	-	-	-	-
ASML >3%	7	42	7.7	3.2	S	-	-	-	-	-
AUML >2%	4	44	7.5	3.3	S	None	-	-	-	-
Bare Soil >3%	6	57	10	9.2	S	T,P	None	-	-	-
BareSoil <3%	6	54	11	8.3	S	S	None	None	-	-
Active Mines	2	56	11	11	S	None	None	None	None	-
Ref. Site	1	78	14	16	T,S	T,P	T,P	None	None	None

Table 7.2 –ANOVA Results for Urban and Coal Mining Land Uses Versus Biodiversity

Land Use/Cover Category	# Sites	Average Data				Significant Group Comparisons			
		Total		EPT					
		(T_T)-Taxa	(T_A)Abun.	(E_T)-Taxa	(E_A)-Abun.	Urban >5%	ASML <3%	ASML >3%	AUML >2%
Urban >5%	2	26	96	2.0	2.0	-	-	-	-
ASML <3%	15	26	117	6.6	37	E_T	-	-	-
ASML >3%	7	20	66	4.6	18	E_A	T_T	-	-
AUML >2%	4	9.0	49	2.0	4.8	T_T	T_T,E_T,E_A	T_T,E_A	-
Ref. Site(s)	1-2	32	208	10	88	T_A,E_T,E_A	E_A	T_T,T_A,E_T,E_A	T_T,T_A,E_T,E_A

Table 7.3 – Results for Minor Strip Mining and AG Land Uses Versus Biodiversity;
(All ANOVA comparisons among 4 categories used in each focus were insignificant)

Focus	Land Use/Cover Category	# Sites	Average Data			
			Total		EPT	
			Taxa	Abun.	Taxa	Abun.
Minor ASML	ASML=0	9	25	104	7.0	37
	0<ASML<3%	4	26	132	6.5	38
AG w/o Urban	Bare Soil <3%	6	26	119	8.2	43
	Bare Soil >3%	7	25	107	5.7	32
Applied to Both	Active Mines	2	29	148	5.0	35
	Ref. Site(s)	1-2	32	208	10	88

Among a mixture of agricultural and urban land uses commingled with strip mines and active and abandoned underground mines, total taxa was found to be a better indicator of degree of AML stress than was EPT taxa. Similar to conclusions that strip mines have significantly degraded aquatic habitat, ASML was also implicated in reducing macroinvertebrate taxon richness. In comparison, abandoned underground mines above stream drainage elevation were found to be more deleterious stresses to aquatic life, requiring only 2% of the aereal landscape to impart significant degradation. Biodiversity was decimated for sites with 10% to 20% AUML. Considering all land uses, conditions where 3% < ASML < 11% were found to impart relatively minor effects under conditions where AUML<2%. The level of ASML needed to impact biodiversity through water quality could not be discerned due to commingled AUML, though effects from ASML up to 15% were not discernable. For remotely sensed geodata, Mine Gob and Mine Spoil data were ineffective in differentiating landscape-habitat and landscape-biodiversity relations. No differences were noted between ASML and unreclaimed ASML.

No significant differences in biodiversity were realized among mining activities in the population of sites with ASML<3%, evaluating criteria for active mines and ASML=0%. Results also indicated no significant differences existed between active mining and agricultural activities with Bare Soil above or below 3% aereal extent. Overall results for macroinvertebrate analysis were most capable in distinguishing significant impact from ASML and AUML. Total taxa data at heavily urbanized sites was not significantly different from mixed agricultural, urban, and active mining entities together representing ASML<3%, and ASML>3%, or reference site data. EPT taxa however showed significantly lower results for Urban>5% compared to reference site data and ASML<3, comparable to impacts seen for AUML>2%. Thus Urban>5%, ASML>3%, and AUML>2 to 10% indicated significant relationships to biodiversity. For ASML>3% effects were attributed to habitat degradation and for AUML>2% to AMD impacts.

7.4.2 Sediment Chemistry and Toxicity

Annual cross-section distributions of sediment chemistry and toxicity are presented in Tables H1 and H2 in Appendix H. Average annual cross-section distributions of sediment chemistry and toxicity are presented in Table 7.4 with dominant land uses in each subshed.

7.4.3 Inter-Test Relationships Among Sediment Toxicity Endpoints

Inter-test results for sediment toxicity endpoints were evaluated for functional significance ($\alpha = 0.05$) and power ($1-\beta$) of regressions. Analysis between sediment toxicity tests indicated no consistent, discernable trends that might explain watershed trends in biodiversity. Inter-test comparisons and comparisons to sediment chemistry were usually insignificant where significant results were weak. Null hypotheses for all inter-annual regressions for individual test endpoints (e.g. *C. tentans* growth in 1996 versus growth in 1997) were accepted ($\alpha = 0.05$) where all but *D. magna* survivorship met tests for normality. For reference to sediment chemistry over the same time period, it was previously determined that inter-annual comparisons for sediment chemistry were strong for Zn and Fe (R^2= 0.71 to 0.74, respectively). Weaker results for Mn and %carbon were found (R^2= 0.59 to 0.51, respectively), and the null hypothesis was accepted for Cu. Inter-annual variation of Cu levels in sediment was attributed to increased agricultural activities where levels were still relatively low compared to crustal abundance.

Station ID	Land Use						Sediment Chemistry					*D. magna*		*C. tentans*	
	ASML	AUML	No.2	No.31	AG	Urban	Carbon	Copper	Zinc	Manganese	Iron	Neonate	Survivorship	Growth	Survivorship
							%	mg/kg	mg/kg	g/kg	g/kg	Count	%	grams	%
(R)FCS1	x	x			x		0.41%	30	75	1.1	43	82	90%	4.0	62%
(R)FCS2	x	x			x		0.37%	55	83	1.0	49	60	92%	1.3	64%
(R)FCS3	x	x			x		-	21	74	1.1	44	13	24%	3.6	78%
(R)SCS1	x	x			x		-	-	-	-	-	58	84%	2.8	47%
(R)SCS2	x	x			x		-	-	-	-	-	17	84%	3.2	88%
LCS1					x	x	1.6%	35	98	1.5	43	81	84%	1.8	35%
LCS2					x		0.65%	15	31	0.64	14	145	92%	2.9	67%
LCS3					x		0.47%	33	24	0.44	9.6	115	90%	1.9	56%
LCS4			x		x		0.17%	15	27	0.35	11	88	90%	3.1	57%
LCS5B	x		x		x		0.30%	19	44	0.69	24	100	90%	2.2	51%
LCS6	x		x	x	x		0.26%	17	36	0.42	16	41	78%	3.6	51%
LCS7	x		x	x	x		0.13%	14	30	0.38	14	29	52%	3.5	63%
LCS8	x	x	x	x	x		0.40%	10	48	0.56	15	61	86%	5.1	73%
LCS9	x	x	x	x	x		0.40%	16	65	1.2	16	33	58%	4.0	60%
LCS10	x	x	x	x	x		0.66%	15	63	0.47	23	48	82%	3.2	79%
TS1					x	x	1.0%	17	54	0.71	27	48	98%	2.4	65%
TS2					x	x	1.6%	21	78	1.1	41	39	96%	2.3	74%
TS3B					x		0.17%	14	30	0.52	14	65	92%	2.2	71%
TS4					x		0.40%	16	25	0.23	10	57	78%	5.6	80%
TS5					x		0.07%	11	16	0.26	7.7	53	62%	1.5	64%
TS6			x		x	x	0.09%	10	31	0.41	18	67	96%	2.6	88%
TS8A	x				x		0.18%	28	41	0.98	18	24	70%	2.5	72%
TS8B	x				x		0.25%	14	50	0.92	16	29	82%	2.3	58%
TS9					x		1.0%	20	55	0.44	24	29	92%	3.1	46%
TS10	x				x		0.86%	37	87	1.5	28	24	78%	3.4	55%
TS11				x		x	0.71%	46	131	2.1	81	29	62%	2.7	83%
TS12					x		0.79%	29	71	1.0	38	17	58%	3.7	60%
TS13	x				x		0.20%	17	32	0.61	17	41	56%	3.3	55%
TS14	x	x			x	x	0.20%	22	72	1.5	16	63	76%	3.7	80%
TS15	x	x			x		0.20%	16	33	0.49	15	61	72%	5.2	69%
TS16	x	x			x		0.38%	20	41	0.44	20	57	72%	4.8	75%
TS17	x	x			x		0.47%	23	80	0.18	39	73	70%	3.5	61%

Table 7.4 – Average Annual Sediment Chemistry and Sediment Toxicity - 1996 to 1997

All inter-test comparisons for combined annual toxicity data failed tests for normality except *D. magna* neonate production versus %survivorship (n = 60; R^2 = 0.25, line slope = $2.59x10^{-3}$, [P]= $4.5x10^{-5}$; α = 0.05, 1-β = 0.99). Significance levels for Spearman-rank correlation coefficients for similar comparisons were all >0.07 except for *D. magna* survivorship versus neonate production r_S = 0.50 (Pearson R^2 = 0.25), and *C. tentans* growth with r_S = -0.38 (R^2 = 0.083), respectively. Presented in Tables H3a and H3b in Appendix H, most inter-test regression comparisons for annual data met assumptions for normality and independence and were insignificant or weak. One exception in meeting test assumptions was *D. magna* neonate production versus *C. tentans* growth in 1997 where normality tests failed due to unusually high midge growth of 9.4 grams at station TS4. Slopes for all annual inter-test regressions were insignificant except for *D. magna* neonate production versus %survivorship (1996: R^2=0.41, [P]=$1.4x10^{-4}$, 1-β = 0.99; 1997: R^2=0.19, [P]=0.017, 1-β = 0.69), and 1996 *C. tentans* growth versus survivorship (R^2=0.26, [P]=0.003, 1-β = 0.88). For the strongest *D. magna* 1996 inter-test results found, the strength of the regression was attributable to 4 of 30 data points. Stations TS11 (below active mine No. 31), TS12, LCS7, and the reference site had low neonate production (13 to 23) and low survivorship (24% to 60%). In 1997, significance in regression results were attributable to poorer neonate responses at TS5, TS13, and LCS9 where poor survivorship was also observed at some sites typified more by agricultural influences.

7.4.4 Landscape-Sediment Toxicity Relations

Table 7.5 presents summary sediment toxicity test results for both years focused on treated and untreated AMD impacts. Based on differentiation seen in sediment chemistry data analyses, site-specific toxicity data for AMD sites were analyzed on two levels in Table 7.5. One approach compared AMD site values to average values obtained for the reference sites in Symmes Creek and Federal Creek. Another compared average values obtained for all Leading Creek sites excluding the AMD sites with AUML>2% and localized impact from active Mine No. 31 at station TS11. Seen in Table 7.5, survivorship was somewhat impacted at reference sites ranging 60% to 84%. Mixed results between years were seen comparing Federal Creek to Symmes Creek. In 1997, the Symmes Creek watershed gave similar survivorship responses but lower *C. tentans* growth and lower neonate production for *D. magna*.

Table 7.5 –Sediment Toxicity Results for Active Mine No. 31 and Sites With AUML>2%

Statistic	Station Group	%AUML	*Chironomus tentans*				*Daphnia magna*			
			Growth (gm) 3.2 (2.7)		%Survivorship 79% (28%)		Neonates (#) 56 (43)		%Survivorship 65% (28%)	
			1996	1997	1996	1997	1996	1997	1996	1997
Federal Creek Avg	(R)FCS1,2,3	0.06%	2.2	6.1	63%	78%	53	78	71%	84%
Symmes Creek Avg	(R)SCS1,2	0.71%	2.8	3.3	60%	75%	-	38	-	84%
Leading Creek Avg	All Sites*	<0.2%	2.4	3.9	76%	51%	63	51	86%	72%
95% Lower C.L.			2.0	3.2	70%	44%	50	30	80%	63%
% of Average Reference Site Data (FCS1,2,3; SCS1,2)	All Sites*	<0.2%	95%	83%	123%	66%	117%	88%	122%	86%
	TS11	0	85%	69%	140%	105%	34%	70%	57%	100%
	TS15	2.8%	199%	113%	143%	65%	95%	122%	108%	81%
	LCS10	4.2%	179%	39%	147%	89%	78%	94%	113%	100%
	TS17	15.9%	100%	95%	134%	52%	81%	179%	91%	90%
	TS16	19.5%	154%	121%	150%	76%	80%	122%	113%	76%
% of Average Leading Creek All Sites* Data	TS11	2.8%	90%	84%	114%	158%	29%	79%	46%	116%
	TS15	0	210%	137%	116%	99%	81%	138%	88%	94%
	LCS10	4.2%	189%	48%	119%	135%	66%	107%	93%	116%
	TS17	15.9%	105%	115%	109%	79%	69%	203%	74%	105%
	TS16	19.5%	163%	146%	122%	115%	68%	139%	93%	88%

**All sites group excludes stations TS11, TS15, TS16, TS17, and LCS10. Pooled average and standard deviation (σ) shown for each sediment toxicity endpoint distribution combining all data results.*

In Table 7.5, TS11 was separately evaluated representing localized impacts from active Mine No. 31. TS11 was found previously to have significantly higher Zn, Mn, Cu, and Fe compared to all sites downstream, as well as compared to all sites uninfluenced by the NPDES discharge. For TS11 data, sediment toxicity tests indicated this site had overall good *C. tentans* growth and survivorship responses for both years, and good production and survivorship were also noted in 1997 for *D. magna*, but not in 1996. In 1996 neonate production at TS11 was low, but low or lower values were also found in one of three samples collected at Federal Creek in 1996, and 1 of 3 samples collected in Symmes and Federal Creeks in 1997. In 1997, relatively low neonate production was found at a number sites in Leading Creek spanning ASML and agricultural land uses. The coal slurry spill at Mine No. 31 in spring 1997 showed no deleterious impact at station TS11 between years in either sediment chemistry data or toxicity data.

Representing the most severely impacted AMD sites, direct examination of raw sediment toxicity data at stations TS16 and TS17 in Table 7.5 indicated robust *C. tentans* growth and survivorship. Relatively good neonate production and survivorship for *D. magna* in both 1996 and 1997 were also observed. The same was true for site TS15 where less AUML impact and weaker AMD water quality signatures were previously documented. Compared to reference site data, *C. tentans* survivorship at TS15 was modestly depressed in 1997 but was unremarkable compared to all sites monitored in Leading Creek. Except for low *C. tentans* growth in 1997, the same was true for the mainstem site LCS10, located just below Thomas Fork (TS17). Midge growth however was very strong in 1996 at LCS10 along with relatively strong survivorship for both organisms tested. For all AUML-AMD impacted sediments, the most noteworthy toxicity test results were slightly lower *D. magna* endpoint values for LCS10, TS16, and TS17 in 1996.

Sites TS1 and TS2 with Urban>5% showed relatively good or above average results for all toxicity endpoints evaluated. Station LCS1 directly below these two sites had good *D. magna* scores and strong *C. tentans* growth. *C. tentans* survivorship though was the lowest of all sites monitored. Direct explanations of impact were not observable in monitoring data at LCS1 except slightly elevated zinc was found at this station. LCS1 zinc levels were just below those seen at station TS11 and near levels seen at other stations. The quarry or small closed landfill might be suspected as sources of this variation where unmonitored toxicants may be involved. A general increase in fine silt/clay sediment fractions can explain elevated results for Fe and Zn.

7.4.4.1 Landscape-Sediment Toxicity ANOVA Analysis

To ascertain generally whether or not unmonitored toxicants might be in part responsible for increased sediment toxicity at some sites, additional ANOVA analysis was completed between metrics of landscape and sediment toxicity. Specifically, groupings established in Table 7.2 and 7.3 were analyzed against individual annual data sets and pooled annual data sets for both years. Parametric and non-parametric tests were used as needed to meet assumptions.

For analogous tests in Table 7.2 comparing dominant urban and AML land uses, only two comparisons were significant and overall results were characterized as weak, inconsistent, or unanticipated with respect to categories evaluated. For *D. magna* neonate production and survivorship all results were insignificant where Urban>5% had the highest overall survivorship averages for 1996, 1997, and combined data. In contrast, Urban>5% and AUML had the overall lowest neonate production in 1996, with Urban>5% typically lower for 1997 and combined data, but AUML with higher output. For *C. tentans* growth significant differences were observed for 1996 and combined 1996-1997 data sets where AUML had higher growth compared to ASML<3% ([P] = 0.0066, 1-β = 0.89; [P] = 0.047, 1-β = 0.45). For this endpoint, individual annual data and combined annual data was generally highest for AUML>2% and second highest for ASML>3%. For *C. tentans*, no significant differences were found. In 1996 well-defined parametric testing for %survivorship was borderline insignificant though elevated response for AUML>2% was borderline significant in non-parametric tests compared to the reference. In this case the smaller group Urban>5% had similar values found for AUML>2%.

For analogous tests in Table 7.3 on dominant AG and minor ASML land uses/covers, few comparisons were also significant. For *D. magna* neonate production all results were insignificant for both the minor ASML and AG focuses. The same was found to be the case for *D. magna* survivorship except for borderline significant increases seen for Bare Soil>3% versus Bare Soil<3%. No significant differences or trends were noted for *C. tentans* growth for both AG and minor ASML focuses. For *C. tentans* survivorship, significant differences were observed for 1997 data and the AG focus ([P = 0.018], 1-β = 0.85). For this comparison, active mines (84%) and the reference site (76%) were generally much higher than Bare Soil>3% (51%) and Bare Soil<3% (42%). For minor ASML focus, results were also significant for *C. tentans* survivorship ([P = 0.016], 1-β = 0.80) where 0<ASML<3% (45%) and ASML<3% (47%) were lower than the stated values for reference data and sites below active mine operations.

7.4.5 Sediment Chemistry Versus Sediment Toxicity

It was previously determined that sediment quality in Leading Creek was within normal ranges of crustal abundance for parameters studied including Cu, Zn, Fe, Mn, and %C. To complete the analytical triangle for sediments between toxicity, land use and chemistry, comparisons between sediment chemistry and toxicity endpoints were also evaluated for functional significance (α = 0.05) and power (1-β) of regressions.

Slopes were negligible for the few cases of significant regressions found between sediment chemistry and toxicity data. For Cu versus log-transformed midge growth regression strength was insignificant (n = 57; R^2 = 0.12, line slope = -5.6×10^{-3}, [P = 0.009]; α = 0.05, 1-β = 0.75). For %C versus log-transformed midge growth, where the null hypothesis was also rejected, regression strength was also insignificant (n = 55; R^2 = 0.08, line slope = -0.11, [P = 0.04]; α = 0.05, 1-β = 0.55). In these tests normality was obtained in residuals with some serial correlation. Evaluating raw and log-transformed sediment chemistry versus toxicity data for 1996 and 1997-1998 events separately resulted in similar graphical relationships. For these regressions null hypotheses were accepted for individual annual comparisons between toxicity tests and sediment chemistry except for 1996 data for *D. magna* versus both Zn and Fe. Individual annual tests all retained normal residuals except for some comparisons involving *D. magna* neonate production and *C. tentans* survivorship. The two 1996 *D. magna* regressions noted with [P] <0.05 had R^2 values between 0.15 and 0.17. In summary, no functional linear or log-linear relationships were observed between sediment chemistry and toxicity tests conducted.

In Table 7.5 average and standard deviation in parenthesis are given for pooled toxicity results (n = 310 for *C. tentans*, and n = 300 for *D. magna*). These statistics combined replicate data for both years and all sites, including TS16 and TS17. Weak correlations between sediment chemistry and toxicity in Table 7.4 were observed in separate pools of data for sites impacted by ASML or AUML. Except for two sites, carbon in sediment was present at levels >0.1% to 0.2% such that toxicant bioavailabilty can be assumed. For stations TS5 and TS6 %C was between 0.07% and 0.09%, but metal concentrations were low compared to other sites in the watershed.

7.5 DISCUSSION

The primary goal of this work was to distinguish between sediment quality and toxicity concerns in mining environments and to shed light on ecotoxicological tests that might be used to assess trends in aquatic biodiversity. Examination of sediment toxicity and its relationship to biodiversity in watersheds with urban, agricultural, and coal mining land uses was constructed along three lines of analysis. These included examination of (1) sediment toxicity inter-test comparisons (2) landscape-toxicity relations, and (3) sediment toxicity-chemistry relations. ASML land uses were previously identified with habitat degradation and not AMD while AUML was related to poor biodiversity associated with AMD, and not habitat degradation. In background analysis completed prior to this work, AMD impacted sites were associated with lower pH and increasing soluble metals. These conditions occurred with increasing AUML extents under increasingly dryer stream flow conditions associated with baseflow discharges.

The range of all sediment concentrations studied throughout the watershed and reference sites were average with respect to crustal abundance and were low with respect to levels that might be needed for toxic response. For sediment toxicity endpoints, inter-test comparisons and comparisons to sediment chemistry were usually insignificant where significant results were weak. It was difficult to assign significance by magnitude of agricultural disturbance or lack of mining influences. The conclusion drawn was that increased active and AML mining activities had generally equal or better response in test organisms than sites in Leading Creek without these influences. This was juxtaposed against opposite trends found in habitat and biodiversity.

7.5.1 Sedimentation, Oxidation, and Streambed De-Acidification

In AMD environments, Fe sinks to sediment as oxyhydroxide flocs and will co-precipitate and adsorb soluble sulfate, enhancing delivery of sulfate to the streambed (Mills *et al*, 1989; Nordstrom, 1982). Mills showed how Fe(III) and sulfate can lead to stable de-acidification of the sediment column through reductive processes essentially reversing pyrite oxidation. Other researcher's results in AML environs have indicated water column acidification does not always result in sediment column acidification. Added to their normal activities of cycling nutrients and mineralizing carbon, the role of benthic anaerobic microorganisms below acidic streams can generate both significant alkalinity and increases in pH, affecting pore-water quality instream (Mills *et al*; 1989). In an AMD impacted stream system in Virginia, Mills and Herlihy observed a sediment pore-water pH of 6.8 within 1 cm of the bed surface when the water column above was pH 4.0. Other research on acidified lakes found a pH of 3.5 in the water column when pore-water within 1 to 2 cm of the bed surface was pH 6.5 to 7.0 (Tremaine *et al*, 1989). AMD does not necessarily lead to *in situ* sediment toxicity.

Stream and lake acidification can lead to increased organic loading to sediments and generally less competition among heterotrophs for organic substrates due to inhibition of aerobic, acid-stressed communities (Rao *et al*, 1989). In Leading Creek, sediments with elevated %carbon levels were attributed to small, heavily urbanized subsheds and were otherwise indifferent to levels of agriculture and mining land uses. While small subsheds were noted, size itself was not a significant factor for elevated %carbon. An inference can still be drawn that conditions for anaerobic activity are enhanced. Mills (1989) listed key biological reactions of de-acidification available instream. Though not applicable to sediments, photosynthesis leads to no net change in alkalinity but increases pH due to removal of CO_2. Nitrate reduction via denitrification will raise pH, and reduction of nitrate respired with nitrite will drop pH due to CO_2 production, but if respired with ammonia will also produce alkalinity. Mn and Fe reduction produce alkalinity in similar but antithetic manner to pyrite oxidation where, similarly, direct and indirect mechanisms may be active. Sulfate reduction will also lead to increased alkalinity and will be greater at lower pH due to H_2S thermodynamics. Finally, amino acid fermentation will increase alkalinity, and methanogenesis can increase pH due to reduction in PCO_2 levels.

Summarized by Mills (1989), surface water acidification processes will often lead to greatly increased alkalinity in beds due largely to reduction of iron and sulfates through biogeochemical reactions taking place in sediments. In the presence of high sulfate, sulfate reducers will out compete methanogenic bacteria, and iron reduction becomes key in re-trapping mineral and hydrogen acidity as sulfides in anaerobic sediment columns. Thus the disposition of Fe(III) iron within the sediment bed remains in a state of flux once precipitation occurs.

7.5.1.1 Sediment Size Effects and Cycled Pyrite Reduction-Oxidation

Exclusive of bacteriological control of pore-water pH, several abiotic factors can also influence mass transfer/transport processes and ultimately biological response in bed framework and matrix materials. These include porosity, permeability, particle size distribution, and time-variant shear stresses instream that determine flushing cycles of armored gravel beds (Burton, 1992; DeWitt *et al*, 1988; Diplas *et al*, 1992; Milhous, 1982; Shirazi *et al*; 1981). Sediment toxicity tests may be influenced by sedimentation and grain size effects (DeWitt *et al*, 1988; ASTM, 1990, 1995). For most macroinvertebrates, domination of bed sediments by particles <2 mm becomes problematic to species survival, and an alternate environment is sought via drift if transportation is available (Allan, 1995; Richards *et al*, 1994a; Vannote *et al*, 1980; Waters, 1995). Potentially perceived as toxicity, displacement effects can also greatly effect bioassemblage measured at a given site. *C. tentans* prefers fine sediment, which is also the particle size that tends to displace large numbers of other macroinvertebrate families in streams (Richards *et al*, 1994a, 1994b).

Whole sediment in-lab toxicity testing strategies that oxidize sediment (Babendreier *et al*, 1997) can mimic deposition-reduction-scour-oxidation episodes instream (Mills *et al*, 1989) and may not accurately reflect *in situ* sediment column conditions. Laboratory-based sediment toxicity tests with sulfides still present in sediment by the time sediment reaches the test vessel can introduce bias, particularly with variable loss or transfer of original pore water. Decreases in diluent pH or alkalinity, and increases in soluble heavy metal species and specific conductivity in the test vessel may occur. Compared to average crustal abundance, sediment chemistry results in

Leading Creek indicated that no adverse ecological response would be expected from concurrent sediment toxicity testing. This assumes that low-concentration sulfide mineral, hydrogen acidity, and toxic metals are not released from *in situ* bed sediments during in-lab testing.

7.5.2 Mine No. 31 Dewatering and Coal Slurry Spill Events

Sulfide oxidation effects have been observed in some testing conducted on southeastern Ohio stream sediments by researchers on this project and in historical studies of Leading Creek sediment (Birge *et al*, 1995). The phenomena was recorded as elevated conductivity and/or decreased pH in test vessel water columns. Initial work conducted by others to assess ecological impacts from sediment associated with the dewatering of Mine No. 31 in 1993 concluded that Parker Run and mainstem sediments were moderately polluted based on similar ranges of sediment concentrations observed here (Birge *et al*, 1995). The referenced study relied on relative comparisons without the advantage of sampling a variety of sites outside the influence of active Mine No.31, ASML, and AUML, both inside and outside of the watershed.

A severe slurry spill occurred from the polishing pond at active Mine No. 31 in spring 1997 due to failure of a discharge pipe within the reservoir berm. The short-term release of fine silt/clay size materials could be observed as a black ring on banks downstream for 30 km to the Ohio River. The residues persisted until rains flushed the system over several weeks. Sediment concentrations downstream of the spill at station TS11 in 1997 though were comparable or lower than 1996 levels. Sediment toxicity after natural flushing was similar or improved. The slurry spilled represented settled solids after AMD treatment and would be expected to be high in Fe, Zn, Mn, and Cu. The dewatering of untreated AMD from Mine No. 31 in 1993 may have contributed in part to overall higher sediment concentrations found on Parker Run in 1996 and 1997. Under normal operations, during initial flushing from rainfall events, Parker Run can turn gray. With active stormwater management in place, clastic/coal dust rinsing from vegetation in the entire subshed was suspected where build-up of dust materials instream may contribute to the hysteresis observed. Analysis indicated these were not concerns for far-field sediment toxicity.

7.5.3 Lack of Sediment Toxicity in Severely Acid Stressed Reaches

It is conceivable that several factors can lead to significant differences between water quality and sediment quality, and associated toxicity tests conducted in Leading Creek and other Appalachian streams. Stark differences between elevated instream water column and low-level background sediment column chemistry for mildly to severely AMD impacted reaches were observed in this study in the concurrent absence of sediment toxicity.

For the three tributary sites classified as severely impacted by AMD and excluding low *C. tentans* growth in 1997 at LCS10, comparing inter-watershed results all toxicity endpoints were within 82% of the lower 95% confidence limit. Except for *C. tentans* %survivorship in 1997 falling to 68% of average reference site data, Leading Creek sites not severely impacted by AMD had similar or better endpoint response (>=80%) compared to the reference watersheds studied. While multiple endpoint analysis among sediment toxicity tests revealed poor consistency, some weak response was noted in *D. magna* results for sites impacted by various mining activities. Unexpectedly good results were obtained for *C. tentans,* indicating better test organism responses at sites with AMD impacts. Results for *D. magna* were slightly lower for AUML impacted sites which might be associated with minor degradation in test vessel water columns due to oxidation of sulfides in sediment resulting from de-acidification.

In areas with abandoned mined lands, *C. tentans* in-lab test responses might increase at sites with poorer diversity, showing possible tolerance for substrates, toxicants, and or acidity (USEPA, 1980). This would be consistent with USEPA's finding for acidified streams due to acid rain. *C. tentans* may also gain advantage found in sandier AML materials with fine silt flocs available at AMD sites, and silts and clays eroded from farmland and banks. The midge's use as an ecological indicator could be potentially positively biased in AML areas, or those areas commingled with AG and AML. Results might inadvertently dilute the perceived impacts of AMD in a risk assessment. For AMD sites, *D. magna* might show negative artifact response with respect to *in situ* conditions where neutral sediment pH actually prevails.

7.6 CONCLUSIONS

Weak to poor correlation was observed between growth and reproduction and associated survivorship, respectively, among and between toxicity tests on an annual basis for both test organisms. For all watershed data, comparisons between toxicity tests among years and within years indicated the modest variation in sediment chemistry present was not a strong determinant in response of multiple test organisms and endpoints. Sediment toxicity testing did not elucidate cause and effect relationships between strong relationships observed previously between urban, ASML, and AUML land uses and poor biodiversity. *C. tentans* response appeared to be positively biased for sites with severe AMD, with inconsistent mild *D. magna* toxic response. Among some agricultural influenced sites, indications were that poor *D. magna* responses were sporadic, alternating among sites with good biodiversity. Classification of agriculture by Bare Soil>3% showed no unusual overall degradation in neonate production. Leading Creek results confirmed a general lack of sediment toxicity and lack of elevated sediment chemistry in reaches severely impacted by AMD. This analysis provided a useful reference point for evaluating the utility of various toxicity tests that might be used to directly assess *in situ* AMD impacts.

In-lab whole sediment toxicity testing was by and large unresponsive to severe AMD impacts. In conclusion sediment toxicity does not appear to be a dominant factor determining lack of benthic macroinvertebrate in Leading Creek. Extremely low biodiversity observed at severely AMD impacted reaches and more moderately depressed biodiversity at other sites in Leading Creek is hypothesized to be a function of water quality degradation. Whole sediment in-lab toxicity tests that use reference water diluent are subject to potential bias and may partially explain the lack of response noted at sites with decimated biodiversity (Babendreier *et al*, 1997). More recently available sediment toxicity testing protocols (ASTM 1995; USEPA, 1994) may offer increased validity and might be more reflective of *in situ* conditions.

Relationships here were not significant for mainstem and tributary stations when comparing sediment chemistry to sediment toxicity and biodiversity scores. Results were not surprising given standard test methods and organisms, AMD stream acidification dynamics, and actual low levels of contaminants monitored in stream sediments. Lack of sediment toxicity associated with sediment chemistry and mining activity impacts have been noted before in severely impacted AMD environs (Babendreier *et al*, 1997; Birge *et al*, 1995; USEPA, 1980). Possibly inhibiting *D. magna* response, sulfides in sediments bound through *in situ* de-acidification might later oxidize in test vessels, invoking a pyretic oxidation-reduction-oxidation cycle. Where water column pH is severely depressed this well-known AMD induced biogeochemical cycle can render *in situ* bed sediments with neutral pH and sufficient alkalinity to buffer infiltrating AMD.

Far-field mainstem impacts downstream from active mine operations might be suspected to be associated with the 1993 AMD dewatering event and an accidental slurry spill in 1997. With the highest relative sediment metal levels found in the watershed, sediment toxicity was not discernable on the receiving tributary. The slurry spill in 1997 also appeared to be of short-term effect where nominal levels or even decreases were subsequently seen in sediment metals and sediment toxicity. Despite findings in 1994 that the dewatering event may have induced moderately polluted sediments downstream on the mainstem, this was not supported in 1996 and 1997 data. Outside of the influence of SOCCO Mine No. 31 and without influence of severe AML-AMD, nearby tributaries were also observed to have modestly elevated sediment metals.

7.7 LITERATURE CITED

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Chapter 8 - ECOTOXICOLOGICAL RISK ASSESSMENT OF URBAN, AGRICULTURE, AND COAL MINING LAND USES USING 48-HR ACUTE WHOLE EFFLUENT TOXICITY TESTING WITH *CERIODAPHNIA DUBIA* AND 35-DAY *IN SITU* CHRONIC TOXICITY TESTING WITH *CORBICULA FLUMINEA*

Justin Eric Babendreier

(ABSTRACT)

AML impact in Leading Creek and its effects upon aquatic biodiversity were previously related to abandoned underground (AUML) mines, and, to a lessor degree to strip mines (ASML), together comprising AML. Underground coal mining operations are still active today where acid mine drainage AMD is treated prior to discharge. Ecotoxicological risk among various land use/cover descriptions was assessed using whole effluent acute toxicity tests with *Ceriodaphnia dubia* and chronic *in situ* toxicity tests with *Corbicula fluminea*. Water column toxicity testing data provided confirmation that for land uses involving AUML>2%, the water column plays a determinant role in pathway exposure of AMD related pollutants associated with the destruction of macroinvertebrate communities. AUML up to 1% by area was found to be relatively non-impacting, and levels between 1.5% and 5% showed moderate impacts, with severe chronic and acute toxicity seen at sites with AUML>10% to 15%. Increasing levels of AUML were consistently tied to AMD and poorer organism responses for acute and chronic toxicity. Mixed results were observed for smaller urbanized subsheds and some agricultural operations. At extreme low flow, instream water quality degradation was seen from treated AMD where increased dissolved solids were correlated to chronic toxicity attributable to elevated chlorides. *In situ* shell length growth rates of *C. fluminea* were determined to be significantly related to drainage area of the catchment, but did not depend on actual flow rates or relative water velocity from site to site. Consideration of drainage area in watershed-scale ecological risk assessments will be required if *C. fluminea* growth rates are to be used to differentiate among a wide array of catchment sizes impacted by water quality degradation.

Chapter 8 - ECOTOXICOLOGICAL RISK ASSESSMENT OF URBAN, AGRICULTURE, AND COAL MINING LAND USES USING 48-HR ACUTE WHOLE EFFLUENT TOXICITY TESTING WITH *CERIODAPHNIA DUBIA* AND 35-DAY *IN SITU* CHRONIC TOXICITY TESTING WITH *CORBICULA FLUMINEA*

8.1 INTRODUCTION

At the end of the 20th century, the U.S. Environmental Protection Agency (USEPA) singled out acid mine drainage (AMD) as the number one water quality problem affecting Appalachia. On watershed scales, defining risk of AMD attributed to coal mining processes will require increased understanding of major processes and environmental mechanisms associated with active mining and abandoned mined lands (AML). In evaluating these major processes and mechanisms, both active and abandoned mining can induce AMD where each can be classified by surface (e.g. strip) mining or underground mining. Hydrogeologic distinctions can also impart useful discrimination between underground mining above stream networks, and mines conducting deeper extraction below streambed elevations. The economic importance of active mining and widespread ecological impact of AML are clear indications that AMD will continue to receive increasing attention and focus in ecological risk assessment. Improved understanding of the interconnections between landscape, mining techniques, and ecotoxicity are essential.

8.1.1 Water Column Driven Toxicity in Coal Mining Environments

Key hydrologic and hydraulic macro-mechanisms are associated with pyrite ores including initial land disturbance, subsequent oxidation, and eventual discharge of acid mine drainage to streams. These mechanisms play pivotal roles in determining how AMD manifests instream as stress to aquatic organisms. Examining the roles of AMD toxicity in sediments and the water column within a hydrologic-hydraulic context will lead to more appropriate ecological risk assessment frameworks in the future. Such frameworks are critically needed today to develop sound management practices for complex ecosystems like watersheds, where both untreated and well-treated AMD might cause toxicity. Differentiating impacts among sources of AMD will frequently be complicated by a hydrologic landscape with commonly occurring impacts attributable to commingled urbanization and agricultural activities.

Ecotoxicity was studied here for its capacity to better understand relationships between biodiversity and various coal mining land uses. Assimilation of the role of water column and sediment columns in risk assessment frameworks was sought in defining characteristics of AMD. This research focused on quantitative assessment of land use/cover and its relation to toxicity and aquatic biodiversity. Complimenting previous landscape-habitat-biodiversity analysis, landscape-toxicity and toxicity-biodiversity relationships were examined. Previous work examined the roles of sediment chemistry-toxicity relationships in impacted AMD environments, quantifying the insignificant role of sediment and associated toxicity *in situ*. This paper sought to improve understanding of related water quality driven ecotoxicological responses to active long-wall underground coal mining, and abandoned underground (AUML) and strip mining

(ASML) activities. Stream hydraulics and commingled urban and agricultural activities were also addressed within the study approach using directly developed and remotely sensed geodata.

8.1.2 Ecotoxicological Framework for Risk Assessment

In the study of landscape hydrology and hydraulics, it is evident that the fate of instream environments depends heavily upon our activities in distinctly unique portions of the landscape, including hillsides, valley floodplains, and riparian zones. An integrated landscape driven ecological risk assessment framework utilized for the study of Leading Creek is depicted in Figure 8.1. To gain critical insight into aquatic ecology and how it relates to the landscape, a wide variety of metrics of watershed health must usually be evaluated (Donigian *et al* 1995; USEPA, 1996c, 1998). This must typically be done within a broad context of conducting multidisciplinary watershed-scale investigations that combine multimedia approaches (Cairns *et al*, 1991, 1996; Donigian *et al*, 1995; Munkittrick *et al*, 1998; Suter, 1998; USEPA, 1996c; 1998). In the framework described in Figure 8.1, ecotoxicological endpoints are intended to reflect ecosystem responses to sediment and water chemistry or other harmful stress. Toxicological endpoints alone though will not usually be sufficient for adequate risk assessment.

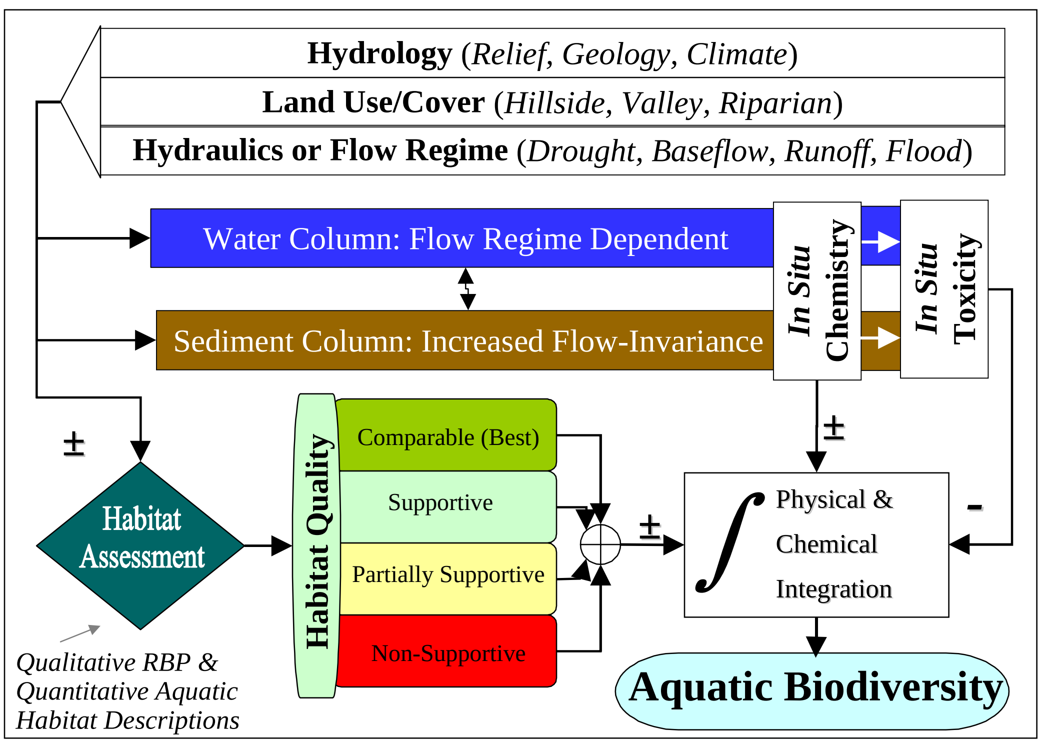

Figure 8.1 – Framework For Hydrologic-Based Watershed Ecological Risk Assessment

The Sediment Triad (Chapman, 1990; Chapman *et al*, 1992) based on sediment chemistry, toxicity, and biodiversity forms the basis of many ecological risk management criteria used in selecting restoration projects for riverine systems (USEPA, 1996c, 1998). Figure 8.1 builds upon Chapman's useful concept, invoking a multimedia, hydrologic-hydraulic approach. It is termed a hydroecological approach here due to the key roles that hydrology and hydraulics play in integrating biotic response. Flow regime plays a profound role in determining how most elements come together instream in determining biotic integrity (Karr *et al*, 1986; USEPA, 1996c; Yoder *et al*, 1996). Biodiversity and sediment toxicity measures augmented with water column chemistry and acute toxicity tests are often primary quantitative elements investigated. Biodiversity as some biometric is more frequently considered the primary dependent variable under investigation (Burton, 1992; Powers *et al* 1992; USEPA, 1996c, 1998). As outlined in Figure 8.1, it is envisioned that contributions from both engineers and scientists are critical to design, implementation, and execution of a well-defined, integrated ecological risk assessment.

Two major limitations exist in current ecological-based risk assessment approaches. First, relationships to stressor sources and related impact are not always well defined nor are they quantitatively identified. Secondly, supporting physically based qualitative habitat measures such as Rapid Bioassessment Protocols (RBPs) may be inadequate in describing variation observed in biodiversity that is not attributed to toxin stress. These concerns lend to an approach that builds confidence on several levels that toxicant sources have been identified and mechanisms of exposure have been clearly defined. The degree to which risk assessment successfully accomplishes these tasks will be proportional to the benefits later derived through manipulation and control of landscapes.

A lingering question is found in the relative role that AMD influenced sediments and water column have upon aquatic biodiversity. A workable ecotoxicological risk assessment framework should: (1) identify the sources of stress from the landscape; (2) identify the primary routes of exposure; and (3) describe underlying physical and chemical mechanisms and processes that can be used to predict the response of indicator biota to the magnitude of stress imparted. Determining quantitative relations between AMD stress to ecology downstream and the magnitude of toxic stress from the landscape was attempted here. Underlying instream mechanisms and processes were previously investigated defining landscape sources and the fate of various minerals and heavy metals in sediment and water columns throughout Leading Creek. In progression, this paper focused on examining the dominant role of water column ecotoxicity.

8.1.3 Toxicity Testing Study Objectives

Experimental design included 10-day in-lab whole sediment tests discussed in Chapter 7, 48-hour in-lab whole effluent acute toxicity tests, and 35-day chronic *in situ* toxicity tests. The toxicity tests were used to assess effects of land use/cover upon aquatic biodiversity defined by qualitative response of benthic macroinvertebrate. The two hypotheses investigated here addressed aspects of hydrologic-based watershed-scale ecological risk assessment by employing GIS based land use/cover analysis. The objective was to evaluate quantitative descriptions of urban, agriculture, surface mining, and underground coal mining activities, and to relate their impacts to ecotoxicity. This paper examined the following hypotheses:

1) Impact upon aquatic ecology attributed to AMD from AML and treated AMD from active underground mining is a function of water quality degradation and not sediment quality.

2) *Corbicula fluminea* (Asiatic clam) growth is known to be dependent upon clam age, stream temperature, and water quality. *In situ* growth rates can also be shown to be spatially dependent where growth rate can be expressed as a function of catchment drainage area.

The two hypotheses were investigated by comparing land use/cover variables to measures of benthic macroinvertebrate biodiversity, and sediment and water column toxicity, respectively. The first hypothesis was initially tested in Chapter 7 by showing that impact upon ecology in Leading Creek attributed to both treated and untreated AMD was unrelated to sediment toxicity. The second hypothesis is partially evaluated here based on a limited data set from watersheds up to 1000km^2 (Babendreier *et al*, 1998). Further reporting is needed to extend hypothesis testing and associated modeling to geographically dispersed watersheds between 1 km^2 and 3x10^6 km^2.

8.2 STUDY WATERSHEDS

Leading Creek is a 5th order stream (scale: 1:24000) in Ohio with 96% spanning Meigs County, and other portions in Athens, and Gallia Counties. Together, 11 mainstem stations and 19 tributary stations were selected for hydrologic, hydraulic, and ecological risk assessment monitoring. The monitoring network is described in Figure 7.1 in Chapter 7. Mainstem stations were located to provide complete longitudinal coverage of Leading Creek, and discrimination among potential sources of sediment and contaminant impacts to the stream system. Various monitoring activities also included reference sites in Symmes Creek and Federal Creek, two other watersheds found in southeastern Ohio. Agricultural activities dominate the upper mainstem of Leading Creek while downstream reaches are dominated by coal mining influences.

8.3 METHODS AND MATERIALS

Data collection in Leading Creek and the reference watersheds included monitoring of: (1) precipitation, stream stage, water velocity, and flow; (2) sediment and water quality; (3) benthic macroinvertebrate and stream habitat; and (4) toxicity through sediment, acute, and *in situ* testing programs. *In situ* toxicity tests included 35-day transplant tests using *C. fluminea*. Stormwater sampling incorporated 48-hour acute toxicity tests on first flush samples using *Ceriodaphnia dubia*. Data collection methods specific to measurement of water flow, water quality indicators, water column toxicity, and land use and land cover are outlined below.

8.3.1 Land Use and Land Cover

Land use/cover data (scale 1:24000) was developed through three sources including a) 7.5-minute quadrangle series maps depicting ASML, b) Ohio Division of Natural Resource (ODNR) geodata depicting AUML, and c) June 1988 Landsat5 Thematic Mapper (TM) geodata developed by ODNR specifically for assessment of abandoned mine lands. This work focused on directly sensed portions of geodata characterizing abandoned mined lands due to its proven strong representation of AML as a significant source of stress to aquatic habitat and biodiversity.

8.3.2 Stream Hydraulics

During the study, ten mainstem sites and six of the tributary sites were included in stream stage monitoring indicated by designation as staff gage stations in Figure 7.1 in Chapter 7. Two continuous gage stations (CGS) at mainstem sites LCS5B and LCS9 were monitored for various parameters on a 5-minute time step. CGS data collection efforts provided continuous tracking of several water quality indicators describing instream environmental conditions throughout the study period. LCS5B is just above Parker Run's confluence and LCS9 is above Thomas Fork's, as shown in Figure 7.1 in Chapter 7. The two continuous gage stations LCS5B and LCS9 reflect significant differences in character between upstream and downstream mainstem landscapes.

Description of hydraulics for toxicity evaluations was important to provide hydrologic inference for acute toxicity testing and the four *in situ* clam studies each completed over a 35-day period (e.g. describing baseflow versus runoff). For 35-day clam tests, time-variant flow conditions will normally prevail within events. Flowrate statistics can also be used to discriminate among instream hydrologic-hydraulic conditions between events. Equipment at each continuous gage station included an ISCO Model-4230 bubbler level meter and Model-675 automatically tipping rainfall gage. Continuous monitoring with automated data storage allowed comparison of stream stage (± 0.025 cm), flow (± 10 to 20%; estimated), and rainfall data (± 0.025 cm). Flow data accuracy is site dependent and flow regime dependent. Stream stage height was calibrated to periodic velocity discharge measurements collected at each staff gage to allow for computation of stage-discharge rating curves (Linsley *et al*, 1975). Flowrates were determined through standard methods employed for stage-discharge rating development at each gage site (Linsley et al, 1975). Details of flow rating are described in Appendices C and D.

8.3.3 Water Quality Indicator Sampling and Analysis

First flush storm grab samples were split for use in chemical analyses and acute toxicity testing. During 1996 and 1997 monthly water column grab samples were collected at routinely monitored sites in addition to collection of periodic stormwater grab sampling and trip and field blanks (APHA, 1992). Field measurements for water column pH, specific conductivity, temperature, and in many cases dissolved oxygen were also completed. Water quality data supporting ecotoxicological testing was previously evaluated in Chapter 6. Not exclusive for all data collected, field meters used included Corning Model-90 Checkmate multimeters, an Orion Model-122 conductivity meter, a YSI-55 oxygen meter, and also a Fisher Accument Model-1003 pH meter. Supplementing site-specific data, equipment at each continuous gage station in Figure 7.1 included a YSI-600 water quality probe installed instream for easy retrieval. Continuous monitoring at CGS stations included pH, specific conductivity, and water temperature.

8.3.4 Acute and Chronic Toxicity Testing

In situ toxicity studies included 35-day transplant tests using *C. fluminea* (Cherry, 1996). In-situ tests were completed simultaneously at all sites during four separate events in spring and late summer 1996 and 1997, respectively. Standard *in situ* sampling procedures consisted of placing five polyethylene (pecan) bags (2.5 mm mesh) at each sampling location. Each bag contained five similarly sized juvenile clams ranging 9 to 14 mm in shell length, with an average of 11 mm (σ = 0.9 mm, n = 2800). Clams were collected from a natural population at an uncontaminated site along the New River, near Ripplemead, Virginia. Clams were measured and individually marked using unique filings on the left and right shells prior to transplant into each study watershed. Clam shell length growth was determined by direct measurement before and after placement using Vernier calipers (± 0.1 mm). Clam survivorship was also recorded. At the bag location, measurements were also collected recording local water depth, and, for two events, local depth-averaged water velocity was taken with a Swoffer Model-2100 velocity meter.

Two acute toxicity tests in spring and fall 1997 were conducted on stormwater run-off grab samples using the first set of grab samples collected after an initial 0.25 cm of rain fell, as best as practical attempting first flush monitoring across the watershed. Screening tests were conducted with *Ceriodaphnia dubia* neonates exposed for 48 hours to various dilutions ranging from 0%, 10%, to 100% composition as site water. A site specific LC_{50} was determined for each undiluted sample showing greater than 50% mortality (USEPA, 1993). Reference diluent water from Sinking Creek in Craig County, Virginia was used and testing was conducted at the Upper Symmes Creek station (R)SCS1. Acute toxicity testing and *in situ* test preparations were performed at the Virginia Tech Environmental Systems Laboratory.

8.4 RESULTS

8.4.1 Land Use and Cover Analysis

ASML mining extents in monitored Leading Creek subsheds ranged from 0 to 30%, and AUML ranged from 0 to 20% by area. Other land use/cover statistics of monitored subsheds are summarized in Table 8.1 addressing urban, agricultural and mining activities. Directly digitized data was derived from aerial photography (ASML) or historic underground mining permit data (AUML). June 1988 Landsat5 coverage was not available for Symmes Creek or Federal Creek. Landsat5 gob and spoil identified near active underground mining areas represented waste rock and coal processing. In previous analysis of landscape relations to habitat and biodiversity, mine gob and spoil were identified in tributaries believed to be not mined and indicated that remotely sensed mining residuals were difficult to distinguish from natural clastic materials not associated with coal mining. Previous analysis incorporated classification of gob and spoil with and without active mining operations. Interpretation of 1988 TM coverage was provided by ODNR.

Leading Creek coverage of abandoned underground mines in Table 8.1 represents 124 non-overlapping coal mine works where all reside above stream drainage elevations. Long wall mining operation aereal extents shown (>100 m below land surface) at Mines Nos. 2 and 31 cover 6300 ha and represent deep groundwater diversion zones associated with treated AMD discharged into Leading Creek. Active mines are dewatered to work the seam and coal is washed. Active mines were not included in AUML totals in Table 8.1. Excluding SOCCO mines, most underground mining in the Leading Creek vicinity was conducted prior to 1910, primarily during the latter portions of the 19^{th} century. Most strip mining occurred after WWII where 80% or 1379 ha of strip mines in Leading Creek were created by 1961, and 20% or 353 ha between 1961 and 1978. Little surface mining occurred in Leading Creek after this period. Quantification of hand digitized ASML based on USGS maps derived from aerial photography, and AUML derived from ODNR mining permit data indicated only 4.2% by area overlap.

8.4.2 Stream Flow and Water Quality Indicators

Figure 8.2 shows *in situ* clam test start dates and instream conditions associated with the two continuous gages at stations LCS5B/CGS1 and LCS9/CGS2. Data is presented for the four 35-day time periods evaluated in spring and late summer of each year for 1996 and 1997.

Table 8.1 –June 1988 Landsat5 TM Data and Directly Developed Geodata on Leading Creek;
(Data in parenthesis gives standard deviation; reclaimed land is applicable to ASML only)

Land Use/Cover Category	Range of Monitored Subsheds	Leading Creek Average
1988 TM		
Mine Gob w/o SOCCO	0 – 5.1% 0 – 0.9%	0.3% (1.0%) 0.0% (0.2%)
Mine Spoil w/o SOCCO	0 – 1.6% 0 – 0.6%	0.2% (0.3%) 0.1% (0.1%)
Bare Soil	0.1% - 8.3%	2.9% (2.3%)
Sparse Veg.	0.3% - 2.7%	1.0% (0.5%)
Moderate Veg.	2.2% - 16%	7.5% (3.0%)
Full Veg.	20% - 51%	30% (6.2%)
Shrub	1.3% - 4.3%	2.3% (0.6%)
Wooded	5.8% - 73%	54.6% (14%)
Water	0 - 2.7%	0.2% (0.5%)
Wetland	0 - 1.1%	0.2% (0.2%)
Urban	0 - 13%	1.0% (2.7%)
Direct Data AUML	 0 – 20%	 1.6% (4.7%)
ASML Unreclaimed	0 – 30% 0 – 30%	3.6% (6.5%) 3.1% (6.4%)

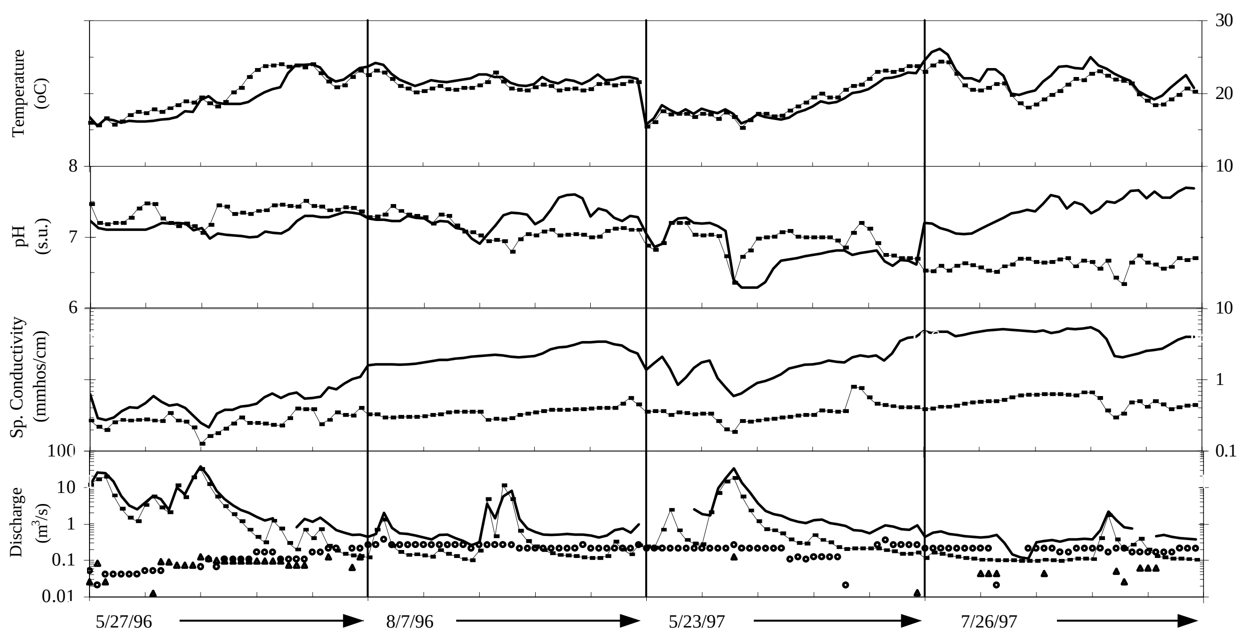

Figure 8.2 – Continuous Gage Station Averaged Daily Data During *In situ* Clam Testing;
(Gaps between events and Ohio River backwater at CGS2 are not shown. Station CGS1 is given by stippled line; station CGS2 by solid line; Mine No. 31 by circles; and Mine No. 2 by triangles. Missing daily active mine data indicates discharge rates < 0.01 m^3/s)

In comparison to active underground mining, minor point source National Pollutant Discharge Elimination (NPDES) discharges given by the WTP, STP in Figure 7.1 in Chapter 7, and sanitary wastewaters from the active mines represent insignificant loading of treated effluents and flow into the stream system. The active mining operations must meet categorical effluent limitations established for coal treatment (coal washing) plants. Coal preparation plant wastewater effluent is high in total dissolved solids (TDS) primarily composed of sulfate, sodium, and chloride where conductivity at SOCCO Mine No.31 typically ranges between 4.0 to 6.2 mmhos/cm. During lower flow conditions the watershed can gain significant alkalinity from deep long-wall mining operations, helping buffer untreated AML-AMD entering the lower mainstem. The closed limestone quarry discharge at the headwaters is also a significant contributor of alkalinity during low and medium flows. For most periods of flow, Mine No.2 wastewaters are conveyed to Mine No. 31 for treatment and storage before final discharge to Parker Run (TS11). Following periods of increased groundwater infiltration after run-off, some process wastewaters are treated and discharged to Ogden Run (TS6).

8.4.3 Acute Toxicity

Results for acute toxicity test samples collected on all first flush stormwater samples 6/26/97 and 10/24/97 are presented in Table 8.2 and in Table H6 in Appendix H. Except for sites summarized in Table 8.2, acute stormwater toxicity test results using *C. dubia* for 10% and 100% concentration ranges indicated 100% survivorship, with some values ranging 90 to 100%. Sites with mortality ≥50% for undiluted effluent were evaluated using 48-hour dilution series tests to arrive at the LC_{50} values shown. Due to historical experience at sites TS16 and TS17, range finding was not conducted and samples were directly analyzed for determination of LC_{50}.

Table 8.2 – 48-Hour Acute Toxicity Test Results for Storm Water Monitoring- 1997;
(All other routinely monitored stations showed >90% survivorship)

Station ID	Survivorship of *Ceriodaphnia dubia* at Dilution						Full Lethal Concentration Test 48-Hour LC_{50}		
	10% Concentration			100% Concentration					
	Jun-97	Oct-97	Average	Jun-97	Oct-97	Average	Jun-97	Oct-97	Average
(R)SCS1	100%	-	100%	100%	100%	100%	-	-	-
LCS7	60%	100%	80%	100%	100%	100%	-	-	-
LCS10	100%	90%	95%	100%	45%	73%	-	84%	84%
TS1	100%	60%	80%	100%	35%	68%	-	5.6%	5.6%
TS3B	100%	100%	100%	55%	100%	78%	None Gen.	-	-
TS6	100%	100%	100%	100%	55%	78%	-	None Gen.	-
TS14	65%	100%	83%	100%	100%	100%	-	-	-
TS15	100%	100%	100%	50%	100%	75%	58%	-	-
TS16	100%	-	100%	0%	-	0%	68%	6.9%	38%
TS17	60%	-	60%	0%	-	0%	37%	19%	28%

In Table 8.2, sites TS16 and TS17 with AUML>15% were consistently identified with acute toxic responses. Sites TS15 and LCS10 with 2%<AUML<5% were also identified with acute toxicity but to a lessor degree and not consistently at TS15 with AUML=2.8%. Site TS1 with previously noted low EPT taxa attributed to Urban>5% was also observed to exhibit moderate toxicity in addition to TS6 below SOCCO Mine No. 2. Water quality data monitored could not discern cause of impacts at TS6 or TS1. Toxicity at site TS6 was not directly attributable to Mine No.2 based on low reported mine discharge of $1.3x10^{-3}$ m^3/s for the period. Local agricultural operations adjacent to the site TS6 may potentially explain observed toxicity due to periodic herbicide application to the riparian zone observed during the study. Those activities were part of an aggressive channelization effort to increase cultivatable area. Like TS6 exhibiting inconsistent acute toxicity between tests, station TS3, upper tributaries of TS14, and localized floodplain areas of station LCS7 are subject to heavy agriculture. Agricultural influences might be expected to be more dominant in spring when toxicity was observed.

Shown in Table 8.2, site TS14 was the only remaining site with an acute toxic response and potentially significant levels of AMD impact with AUML=1.7%, falling just below the 2% criteria established here for AUML. Like station LCS10 with a minimum pH 6.9 s.u., station TS14 did not exhibit signs of pH depression during the study with a minimum pH value found of 6.8. Several smaller tributaries upstream on Little Leading Creek exhibited severe AMD impacts with minimum pH of 3.9 to 4.0, similar to minimum values found at station TS17 on Thomas Fork. Sites LCS9 and the upper Symmes Creek station (R)SCS1 consistently showed no acute toxic responses. Next to Little Leading Creek (TS14), LCS9 and the reference site (R)SCS1 had the next highest levels of AUML in the study, 0.93% and 0.97%, respectively.

8.4.4 *In situ* Clam Toxicity

In situ toxicity testing results using *C. fluminea* are presented in Table 8.3. Average daily water temperature among all sites and average initial shell lengths (ISL) are also given. Raw data distributions for each test event are presented in Table H6 in Appendix H. Statistics excluding all dead clams among sites with and without severe impact were considered biased. Common practice sometimes excludes dead clams from growth statistics. This can bias comparative land use analysis for growth endpoints when lethal stress is present. Calculated average growth was determined here by assuming zero growth for dead clams. Dead clams were excluded from growth statistics, however, in cases where site mortality was below 20%. Supporting this approach, healthy population mortality of up to 8% to 27% for waters considered non-impacted have been observed by other researchers (McMahon *et al*, 1986; Joy, 1985; Welch *et al*, 1984).

In Leading Creek, moderate, significant correlations were observed between clam survivorship and clam growth for each sampling event. Pearson correlation coefficients for the four events conducted ranged in chronological order 0.74, 0.63, 0.78, and 0.83, with the average distributions exhibiting a correlation coefficient of $R = 0.75$ ($1x10^{-6} = [P] < 4.6x10^{-4}$). Spearman-rank correlation coefficients for each event were slightly stronger (0.84, 0.67, 0.85, 0.84, respectively) indicating in general fair agreement found between survivorship and growth rate endpoints for most tests conducted. Inter-endpoint analysis also indicated strong functionality between log-transformed growth and survivorship in three of four cases. For example in fall 1997, a calculated $R^2 = 0.81$ was observed among non-zero data.

Similar to sediment toxicity evaluations in Chapter 7 using urban and AML landscape geodata, ANOVA comparisons were also analyzed here for relations between landscape influence and clam toxicity responses. Similar to trends seen in spatial dependence discussed further below, for the four clam toxicity test events and averaged data across events, significant results were obtained only for June 1996 growth and survivorship and August 1977 growth (α =

0.05). In the case of data averaged across all four events, inclusion of LCS10 led to insignificance in results for the group AUML>2%. Characteristically, land uses/covers described by AUML>2% and to a lessor degree Urban>5% exhibited poor clam responses compared to other categories. Overall trends in clam response mimicked acute toxicity results and results previously found between biodiversity and landscape summarized in Chapter 7.

Table 8.3 – *In Situ* Clam Toxicity Results - 1996 to 1997; (For all sites, average and standard deviation in parenthesis are given for average daily water temperatures and initial shell lengths)

Station ID	35-Day Event Averaged Clam Growth (mm) and % Survivorship									
	June 1996		August1996		June 1997		August 1997		Average	
	(mm)	%Surv.	(mm)	%Surv.	(mm)	%Surv.	(mm)	%Surv.	(mm)	%Surv.
(R)FCS1	0	0%	0	0%	0.1	36%	-- No Sample --		0.0	12%
(R)SCS1	-- No Sample --		-- No Sample --		0.6	92%	1.5	96%	1.1	94%
LCS01	0.8	88%	0.5	76%	1.1	96%	0.9	100%	0.8	90%
LCS02	0.7	48%	0.2	16%	0.9	96%	1.0	88%	0.7	62%
LCS03	1.2	80%	0.3	36%	1.5	100%	1.3	100%	1.1	79%
LCS04	2.0	84%	0.2	12%	0.2	20%	2.1	100%	1.1	54%
LCS05B	0.7	40%	0.2	24%	0.1	30%	1.7	100%	0.7	49%
LCS06	2.4	100%	1.7	88%	0.0	20%	0.7	78%	1.2	72%
LCS07	2.5	100%	0.3	24%	0.9	84%	1.3	92%	1.3	75%
LCS08	2.3	76%	1.2	76%	0.0	0%	1.9	100%	1.4	63%
LCS09	3.1	100%	1.8	76%	0.2	96%	1.8	100%	1.7	93%
LCS10	0	0	0.6	28%	-- Lost --		1.9	100%	0.8	43%
TS01	0.2	44%	0	0%	0.5	72%	0.2	68%	0.3	46%
TS02	0.2	52%	0.4	60%	1.4	100%	0.4	33%	0.6	61%
TS03B	0.2	16%	0.3	16%	1.3	100%	0.1	29%	0.5	40%
TS04	0	0%	0.4	32%	0.3	20%	1.1	92%	0.4	36%
TS05	0.2	20%	0.4	48%	-- Lost --		0.3	48%	0.3	39%
TS06	1.4	84%	1.6	96%	0.6	84%	1.2	100%	1.2	91%
TS07	0	0%	-- No Sample --		-- No Sample --		-- No Sample --		0.0	0%
TS08A	0.1	12%			0.0	0%	-- No Flow --		0.0	6%
TS08B	0	0%	0.3	48%	0.0	0%	1.6	100%	0.5	37%
TS09	0.7	72%	0.6	80%	0.9	100%	1.5	100%	0.9	88%
TS10	1.0	96%	1.2	64%	0.1	32%	1.0	100%	0.8	73%
TS11	1.5	88%	2.1	72%	0.1	44%	0.2	48%	1.0	63%
TS12	0.6	56%	0	0%	-- Lost --		1.1	100%	0.6	52%
TS13	0.1	100%	0.1	96%	0.4	84%	0.3	79%	0.2	90%
TS14	0.1	84%	0.8	92%	0.2	100%	1.5	100%	0.7	94%
TS15	0	0%	0.1	92%	-- Lost --		-- No Flow --		0.0	46%
TS16	0	0%	0	0%	-- Lost --		0	0%	0	0%
TS17	0	0%	0	0%	0.0	0%	0	0%	0	0%
Water (^{o}C)	21.6	(1.1)	22.7	(1.0)	20.5	(1.1)	22.4	(1.0)	21.8	1.0
ISL (mm)	11.8	(0.83)	11.4	(0.84)	10.6	(0.75)	11.0	(0.61)	11.2	(0.51)

For ANOVA comparisons among survivorship and growth endpoints, similar trends were observed across each event and averaged data. These trends followed lowest responses in growth and survivorship in order of AUML>2%, Urban>5%, reference data, and mixed ranks for ASML>3% and ASML<3%. For averaged data sets for the stated order in ranks, 35-day growth was 0.22 mm, 0.45 mm, 0.55 mm, 0.76 mm, and 0.78 mm, respectively. Average survivorship was 22%, 54%, 51%, 65%, and 59%, respectively. In looking at minor ASML activities against ASML=0 and active mining operations, significant results were found for the June 1996 event where ASML=0 had lower growth (0.56 mm) compared to 0<ASML<3% (1.86 mm). The only other significant result observed for the minor ASML focus was for August 1996 data where active mining sites showed significantly higher growth than other categories. In examination of focus placed on agriculture via Bare Soil geodata, the only significance found among comparisons of event and averaged data were similar results for active mining in August 1996.

8.4.4.1 *In situ* Clam Event Test Conditions

Continuous monitoring station data for CGS1 and CGS2 are summarized for each test event in Table 8.4. The data exclude backwater effects from the Ohio River and days with missing CGS data, where missing rainfall data at CGS1 was supplemented with NPDES data from active mine operations. Temperature data shown in Table 8.3 and Table H5 in Appendix H were derived from average 35-day temperatures modeled at each site based on linear regressions with instantaneous data at station CGS1 and monthly grab sampling data at each site conducted between 1996 and 1997. Temperature models across all sites monitored had an average R^2 value of 0.94 (σ = 0.04). Except for site (R)FCS1, linear temperature model R^2 values ranged between 0.89 to 0.99 (n = 11 to 16). Similarly strong correlations were also realized using CGS2 data.

Table 8.4 – CGS Station Instream Conditions During *In Situ* Clam Toxicity Tests

Event	Event Statistic	CGS1/LCS5B at Dexter Ohio				CGS2/LCS9 above Middleport, Ohio			
		Jun-96	Aug-96	Jun-97	Aug-97	Jun-96	Aug-96	Jun-97	Aug-97
Rainfall (cm)	Total	15.7	10.6	12.6	12.7*	10.9	-	18.8	11.6
Flow (m^3/s)	Minimum	0.12	0.10	0.14	0.09	0.46	0.22	0.47	0.10
	Average Daily	4.9	0.83	1.8	0.19	6.9	1.1	3.9	0.50
	Maximum	35	17	25	3.2	42	14	38	3.1
Flow Condition (# days)	Low Flow	8	28	21	33	0	5	0	16
	Medium Flow	10	4	7	2	12	26	19	18
	High Flow	17	3	7	0	21	4	10	1
pH (s.u.)	Minimum	7.0	6.7	6.2	6.2	7.0	6.3	6.2	6.9
	Average Daily	7.3	7.1	6.9	6.6	7.1	7.3	6.8	7.4
Spec. Cond. (mmhos/cm)	Average Daily	0.27	0.35	0.37	0.50	0.51	2.32	1.69	4.18
	Maximum	0.49	0.59	1.00	0.69	1.17	3.51	4.08	5.53
Water Temperature (^oC)	Minimum	14.6	19.6	13.7	17.6	14.7	19.4	14.4	15.8
	Average Daily	20.0	21.2	18.8	20.9	19.3	22.0	18.5	22.4
	Maximum	24.9	24.1	24.3	25.1	25.4	25.1	23.3	28.5

Shown in Table 8.4 and Figure 8.2, flow was highest in spring each year, and spring water temperature was 2-4 ^oC colder. Optimum temperatures for clam growth are expected to range between 21-23 ^0C, indicating some effects in growth rates from spring to fall might be expected (Foe *et al*, 1986a; Mattice *et al*, 1985). Average temperatures were for the most part above 20 ^oC. Differences due to ambient temperature variation between events and between sites during events would not be expected to dominate variation induced by other environmental factors present in Leading Creek. A significant factor for sites below Mine No. 31 was noted in total dissolved solids that varied significantly for each of the four events. This was observed in specific conductivity data at CGS2 and grab data at other stations. For the August 1997 event, station LCS9 specific conductivity averaged 4.2 mmhos/cm and rose as high as 5.5 mmhos/cm.

Disparity between flow condition classification at CGS1 and CGS2 in Table 8.4 is due to NPDES discharge inputs at Mine 31. Relatively consistent for normalized unit drainage area data, CGS1 flow condition reflects sites uninfluenced by Mine No. 31. As shown in Figure 8.2

and Table 8.7, the June 1996 event was characterized early on by high flow run-off followed by periods of medium and low flows, respectively. The following two events were characterized primarily with low flow conditions with two interspersed pulse flows, with the second set in spring 1997 experiencing greater magnitude and longer duration in run-off characteristics. The fourth *in situ* test event was characterized by low flow that led into later drought conditions in September and October. Shown in Table 8.3, clams were not placed at two sites due to dry stream conditions. *In situ* testing covered a wide variety of run-off conditions and baseflow.

8.4.4.2 Spatial Dependence of Clam Growth Upon Drainage Area

A strong dependency was found between *C. fluminea* growth rate and spatial scale but not flow rate measured during the test conditions. Examining spatial dependence of clam growth data, Figure 8.3a and 8.3b shows the relationship found in Leading Creek for growth rate data at sites not impacted by AMD, defined by sites with AUML<2%. Previous work in Leading Creek showed a clear distinction between this land use/cover category and soluble metals in the system, particularly for zinc and manganese. For AUML>2% and particularly for AUML>10% elevated metals often tracked increasing sulfate and low pH under most low and medium flow conditions. Owing to their utility in risk assessment, it is known that lethal and sub-lethal responses in clam endpoints can be attributed to variation in water quality, among other factors (Belanger *et al*, 1986, 1990; Fritz *et al*, 1986). Because clams are known to respond negatively to zinc, an evaluation of segregated data using AUML>2% was initially appropriate in attempting to consider natural non-impacted *in situ* clam growth response and potential spatial dependencies.

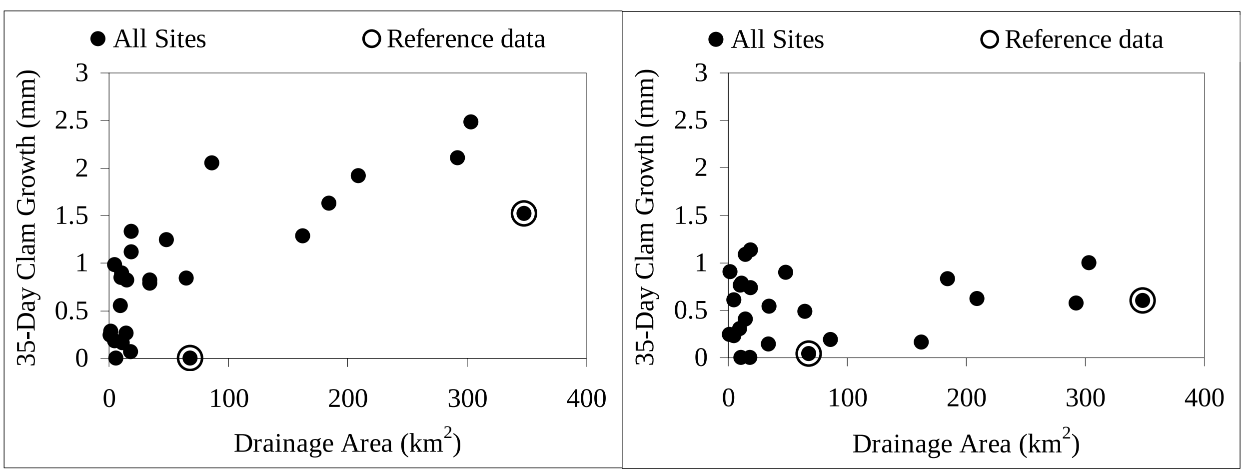

Figure 8.3a and 8.3b – Averaged 35-Day *C. fluminea* Growth versus Catchment Drainage Area for (a) June 1996 and August 1997, and (b) August 1996 and June 1997

In Figure 8.3b, the two events most characterized by moderate pulsing storm flows in Figure 8.2 and Table 8.4 elicited poorer clam response at larger sites where AUML<2%. This may indicate potential sensitivity of the species to additional water quality toxicants associated with non-AMD impacted landscapes such as agriculture. In non-parametric ANOVA testing of averaged low flow and high flow dominated data shown in Figure 8.3a, a significant difference was found between sites above (1.85 mm) and below (0.64 mm) a threshold criteria of 75 km^2 catchment size ([P]= 0.00175, 1-β =1.00). Looking at individual clam test events, similar drainage area dependence was found in June 1996 (2.17 mm versus 0.50 mm; [P]=0.001, 1-β =1.00) and in August 1997 (1.58mm versus 0.86mm; [P]=0.0062, 1-β =0.83). The June 1996 event met parametric test assumptions ([P]=$3x10^{-6}$) where averaged data and the event for fall 1997 event were borderline rejected for kurtosis normality.

The spatial dependency effect observed in Leading Creek clam growth was most readily recognized traveling down the mainstem from headwaters to the creek mouth. The Federal Creek site that showed consistently poor growth responses was omitted from this analysis to avoid bias. The reason for poor response at this reference site could not be discerned from available data. Based on low AUML extent of 0.06%, significant AMD or AML would not be suspected. The more responsive reference site on Symmes Creek had higher ASML and AUML.

8.4.5 Comparison of Acute and Chronic Toxicity Test Results

Figure 8.4 gives summarized average results for key land use/cover categories. In comparing acute versus chronic results, severely impacted clam responses were noted at one of the reference sites, affecting average data shown. The cause of impacted clam responses at the Federal Creek site could not be discerned from available water quality data. The degradation in organism response seen would not be expected to be associated with coal mining activities based on land use/cover data. For each test, Figure 8.4 showed increasingly greater ecotoxicological stress with increasing amounts of AUML, supporting previous land use/cover and chemical analyses. Results for sites with AUML>2% could be directly attributed to increasing zinc toxicity. Urbanization in small subsheds was seen as a more sporadic, but still potentially significant concern for impacted water quality and toxicant loading. In cases with increased urban land uses, the specific toxicant stress was not directly identified.

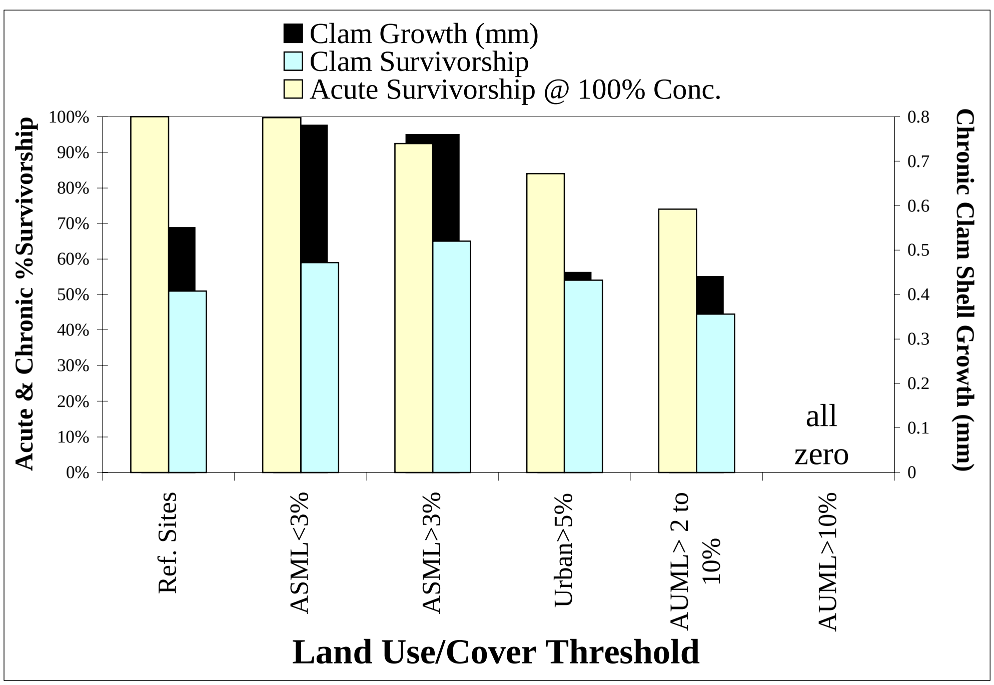

Figure 8.4 – Acute and Chronic Water Column Ecotoxicological Test Results

8.5 DISCUSSION

Together with discussion in Chapter 7 on sediment toxicity, this paper's primary goal was to further distinguish between water and sediment quality concerns in mining environments and to shed light on the utility of various ecotoxicological tests. Water quality driven ecotoxicity and its relationship to land use/cover and biodiversity in watersheds with commingled urban, agricultural, and coal mining land uses were examined through acute and chronic water quality based ecotoxicological testing. This included evaluation of first-flush acute stormwater run-off using *C. dubia,* and *in situ* chronic toxicity testing using the bivalve filter feeder *C. fluminea*.

8.5.1 Land Use/Cover and Its Relation to Water Quality Toxicity

Analysis of acute and chronic water quality toxicity assessments both indicated consistent results implicating AMD from AUML as the most dominant stressor in Leading Creek. These results tied in directly with previous findings examining sediment and water chemistry, and habitat and biodiversity depicted in Figure 8.1. One urban site (TS1) and one agriculturally impacted site (TS3) also had low biodiversity scores. These two sites exhibited poor clam growth, clam survivorship, and acute toxicity response, but otherwise satisfactory sediment and water quality tests. Unmonitored water quality degradation from pesticides, nutrient loading or pulse dosing of heavy metals will not always be captured in discrete water quality monitoring cycles, particularly when associated with run-off. This dynamic facet of watersheds highlighted benefits gained from an integrated multimedia, multidisciplinary approach to ecological risk assessment. The role of ecotoxicological testing was found to be an integral element in soundly inferring relations between biodiversity and land use/cover. In this case, water quality dominated concerns.

8.5.1.1 Acute Water Column Toxicity

Lower flows and mild run-off flushing the landscape characterized water quality obtained during two stormwater sampling events in June and October 1997. In these sampling events only limited dilution of run-off occurred with modest increases in stream stage noted, ideal for acute toxicity assessment. Acute toxicity testing data provided confirmation that for land uses involving AUML>2%, the water column played a determinant role in pathway exposure of AMD related pollutants associated with destruction of macroinvertebrate communities observed. Data collected inside and outside the Leading Creek watershed indicated that ranges up to 1% AUML were relatively non-impacting. AUML land use levels between 1.5% and 5% by area showed moderate impact with severe acute responses seen at sites with AUML> 15% in aereal extent.

8.5.1.2 Chronic *In Situ* Clam Toxicity Testing

Examining Table 8.3, AUML was differentiated by separation of groups at 2% and 10% AUML. Consistent AUML impacts upon clam growth and survivorship responses were consistently observed at tributaries TS15, TS16, and TS17 with AMD impacts. The exception was station Titus Run (TS15) with AUML=2.8% and milder AMD where growth was typically zero or nil but survivorship was sometimes fair or good. Overall, threshold analysis for AUML and urban land uses were identified under various events and flow conditions as having significantly poorer clam survivorship and growth responses compared to ASML. There was, however, a general lack of strong correlation in direct comparisons of biodiversity data and clam response data (R^2=0.01 to 0.28 for survivorship, and R^2=0.00 to 0.19 for growth data). This can be explained in part by the differentiated nature of biodiversity in responding to various types of anthropogenic influences occurring on the landscape (e.g. EPT taxa were sensitive to urbanization, but total taxa were not).

Active mining was shown in Leading Creek to be a significant factor related to impaired clam growth and survivorship during lower flows. In fall 1997, effluent loading was observed to cause impairment to *in situ* clam growth throughout Parker Run and on the mainstem below its confluence, apparently attenuated downstream by dilution. This coincided with the lowest natural flows encountered during four clam test events conducted. Degraded growth response was observed when active mine effluents dominated instream flow. For the August 1997 event this was correlated to average instream specific conductivity levels approaching 4 to 5 mmhos/cm, becoming increasingly deleterious above these levels. This aspect was believed to have dampened normally stronger growth dependency upon catchment size at those sites included in Figure 8.3a.

8.5.2 Chronic *In Situ* Toxicity Testing with *C. fluminea*

Somewhat less well known, *in situ* toxicity testing using *C. fluminea* has been identified as a tool for conducting ecological-based risk assessment. Other techniques used here for sediment and water column evaluation represent more familiar bioassay approaches. *In situ* transplant techniques using caged individuals have been applied successfully to over twenty watershed systems and estuaries using a variety of growth-rate evaluations. Shell length growth measurement techniques range from periodic size-frequency analysis of natural populations, direct “before and after” growth measurement of caged or “release and catch” individuals, and finally a less studied technique counting incremental growth rings on the shell (Fritz *et al,* 1986; McMahon *et al,* 1986). *C. fluminea* can easily be obtained from an unpolluted reference stream and introduced *in situ* to provide survival and growth data for polluted or impacted sites (Belanger *et al* 1985, 1986, 1987, 1990; Farris *et al* 1988, 1989, 1994; Graney *et al* 1980, 1983, 1984). Previous testing with *C. fluminea* in reference and impacted streams has shown that variation can be observed in growth rates of caged individuals at a single site or at a few sites along a reach. Investigation of Leading Creek extended use of *C. fluminea* to watershed-scale and subwatershed-scale assessments.

Lethal and sub-lethal responses in C. fluminea have been attributed to variation in water quality, among other factors (Belanger *et al*, 1986, 1990; Fritz *et al*, 1986). *C. fluminea* sensitivity to elevated heavy metal concentrations instream has been successfully demonstrated in field-based artificial streams, *in situ* conditions, and lab conditions (Belanger *et al*, 1986, 1990; Cherry, 1996). The Asian clam responds positively to nutrient or organic material enrichment and is relatively independent of siltation effects in the water column (Belanger *et al*, 1985; Foe *et al*, 1985). Application of *C. fluminea* as a test organism in ecological risk assessment have included studies of active and abandoned mined lands (AML), and industrial, and municipal discharges involving organic and metal wastes (Cherry, 1996; Fritz *et al*, 1986). The Asiatic clam is sensitive to a wide variety of environmental stresses. *C. fluminea* has been shown to respond negatively to water quality impacts associated with hypoxia, coal ash slurries, heavy metal toxicity with Zn and Cu, halogenation (e.g. chlorination), and is also a particular nuisance to power plant thermal operations and natural stream ecology (Belanger *et al*, 1990; Cairns *et al*, 1983; Doherty, 1990; Foe *et al*, 1987;

McMahon *et al*, 1986). Asian clams borrow in sediments, are able to travel at rates up to 250 cm/hr, and prefer coarse sand material for burrowing (Belanger *et al*, 1985).

Acute stormwater testing was successful in identification of water quality problems at many sites but represented only an instantaneous snapshot of pollution. The 35-day *in situ* clam test is capable of integrating variable chronic toxic exposures occurring throughout the test duration, capable of recording growth increments and system disturbances on a daily basis (Fritz *et al*, 1986). *C. fluminea* is suited well for AMD and agricultural cycles driven by a pulsing climate and subsequent hydrologic assimilation of pollution that delivers toxicants to the stream system through run-off and baseflow discharges. Because of the ability of Asian clams to depurate heavy metals and other contaminants in a matter of days to weeks, bioaccumulation studies using *C. fluminea* would not appear to be feasible for monitoring acute or chronic responses to contaminants instream.

Foe and others have shown that using a condition index relating visceral mass to shell mass is also a limited test strategy for ecological risk assessments due to seasonal fluctuation in the metric attributed to reproduction (Foe *et al*, 1987). Bioaccumulation metrics could though be strategically used to augment *C. fluminea* shell growth if specific contaminant identification strategies were sought. Due to the exploitable mechanism of *C. fluminea* to “clam-up” in poor water quality, causing slower growth, it does not appear to be a viable candidate for acute exposure tests less than 7 to 14 days. Degrowth associated with induced chemical stress has been shown to occur for optimal warmwater temperatures, thermally inhibited waters, and dormant winter periods of cold water. Exceptions for shorter-term acute exposures using *C. fluminea* include studies in dissolved oxygen sag, salinity inhibition, or thermal inhibition which are strong factors affecting survivorship of the Asian clam (Foe *et al*, 1986a, 1986b; McMahon *et al*, 1979; Mattice, *et al* 1985).

8.5.2.1 Environmental Factors for *In Situ* Clam Toxicity

All sites in Leading Creek met nutrient saturation values > 80% with most above 90% of expected optimal phytoplankton growth, indicating N and P were not limiting factors in phytoplankton growth. Growth limitation factors were derived from a standard Michaelis-Menton nutrient-limitation model with assumed half-saturation constants of 0.015 mg/L for N and 0.0025 mg/L for P, respectively (Thomann *et al*, 1987). Normally stream flow transport rates may limit cell growth reproduction rates (Allan, 1995; Thomann *et al*, 1987).

In total, four separate *in situ* 35-day clam tests were conducted. Sampling examined a broad range of flows during the testing, covering low-flow to bankfull flow regimes. During *in situ* clam testing, continuous flow, temperature, and water quality data were automatically monitored at the two continuous hydrologic stations. Watershed flow models developed were used to generate average flow rate, minimum flow rate, maximum flow rate, total flow, and the number of storm pulses expressed for each event. With respect to *C. fluminea* shell-length growth rates determined *in situ*, mean, minimum, maximum, and total stream flow rates were evaluated and combined for analysis over multiple test events. Water velocity data was also evaluated. *In situ* growth rates of *C. fluminea* appeared to depend strongly on drainage area of the catchment but not on actual flow rates or relative water velocity from site to site. Because drainage area can be used to describe relative flowrates from site to site (Leopold *et al*, 1964), growth rate dependence on catchment size is hypothesized to represent a surrogate for food supply and phytoplankton dynamics (Foe *et al*, 1985; Minshall *et al*, 1985).

Supplemented with formal landscape and water quality assessment completed in the interim, this work provided a clearer distinction and justification for segregation of sites based on water quality than an initial analysis performed on *C. fluminea* growth rates *in situ* (Babendreier *et al*, 1998). In that work, a Monod-type model for predicting non-impacted growth rates as a function of drainage area was fit to Leading Creek data (maximum growth = 0.07 mm/day, Ks = 2700 hectare @ 20 ^{o}C, $R^2 = 0.7$). While the model approach worked well for moderate sized watersheds, it may not address significantly larger systems >1000 km^2. More robust parameter estimates may also be better suited for applications across multiple ecoregions. A full examination of spatial dependence is still needed. This dependence was formally established here as a trend of great importance affecting the efficacy of use of shell length growth rates in ecological risk assessment strategies. Consideration of drainage area in watershed-scale ecological risk assessments will be required if *C. fluminea* tests are to be used to identify sites potentially impacted by water quality degradation across a wide array of catchment sizes.

8.5.3 Multimedia Triad Confirmation in Leading Creek

Figure 8.1 depicts a multimedia ecological risk assessment framework for investigation of watershed environments. The framework was based on a basic triad approach in relating chemistry and toxicity to biodiversity within a supporting hydrologically driven habitat and landscape context. Stark differences between instream water column and sediment column chemistry for mildly to severely AMD impacted reaches were observed in our study in the absence of sediment toxicity. In this risk assessment, sediment and water quality data supported findings in ecotoxicity data. With previous findings summarized in Chapter 7 relating habitat and aquatic biodiversity to landscape influences, data investigated here supported the conclusion that water quality toxicity was significant in Leading Creek and was strongly associated with concerns previously identified with urbanization in small subsheds and abandoned underground mined lands (AUML). It was also determined that sediment chemistry and sediment toxicity concerns were minimal and comparatively insignificant.

Like Chapman's sediment triad, a pivotal assumption in applying the multimedia triad is that each of the three corners measured is accurate and representative of variation of interest at each site. If biases or other confounding factors invalidate the sediment toxicity test for example, then conclusions from the multimedia triad will be less strongly held. Efforts underway to simplify toxin impact assessment through single-component analyses are important, i.e. use of the sediment toxicity portion of the triad (Chapman, 1995; Long *et al*. 1996; USEPA, 1995, 1998). Reliable and accurate chronic toxicity tests will be essential to meeting the underlying goal in this trend which will continue to expand in applications of ecotoxicity and biometrics. There is more work to be done in the area of toxicity test development and continued verification of the promising state of the art found within *in situ* testing regimes (Cherry, 1996). *C. fluminea* is believed to show great promise, though clearly a more developed understanding of environmental factors affecting growth and survivorship *in situ* is needed.

Both acute toxicity and chronic toxicity investigations of water quality concerns were substantially supportive of each other in assessment of Leading Creek. The two approaches both identified similar concerns in land use/cover geodata that pointed to urban and AUML influences, as did previous direct analysis of landscape-biodiversity relations. The work lead to a final conclusion that strip mining was problematic to habitat degradation, AUML was deleterious to water quality degradation, and urbanization in small subsheds affected both water quality and habitat. For active mining, agricultural influences, and minor strip mining, impacts were difficult to consistently observe. Risks presented to ecology by these activities in Leading Creek imparted some variability in data sets and may show significant influence upon biota instream at some level, viewpoint, and cross-section of the landscape, though not fully realized here.

8.6 CONCLUSIONS

Together, water and sediment chemistry data supported conclusions that within coal mining environments, sediment toxicity was not a dominant stressor at even the most severely AMD impacted sites where water column pH reached 3.7. The underlying data set examining sediment chemistry and sediment toxicity included sites without mining but under influence from urban and agricultural practices, sites with a mixture of influences from active underground mining, abandoned underground mining (AUML), and reclaimed and unreclaimed abandoned strip mines (ASML). Focus upon water quality based ecotoxicity and the water column identified this route of exposure as the greatest concern. Aereal extents of AUML>2% to 10% and urban activity >5% were tied to poor organism responses for both acute and chronic toxicity testing conducted here.

Multiple toxicity tests were evaluated for their relative ability to identify water quality degradation, and for their strength to indicate upstream mining activity and type. Of three toxicity tests used, including sediment, acute water column, and *in situ* methods, acute and *in situ* transplant testing with *C. fluminea* were reliable indicators of acute and chronic water quality degradation and reduced biodiversity. Permitted discharges for treated AMD and untreated AML-AMD were both shown to cause lethal and sublethal responses in clams placed *in situ*, with results depending on instream flow condition. In Leading Creek, lethal and sublethal clam response instream was attributable to variation in water quality associated with zinc and possibly other metals for untreated AML-AMD. Reduced growth patterns and survivorship were observed in most subsheds impacted by AMD, on both mainstem and tributary reaches. For treated AMD from active mining, increasing TDS loading instream during lower flows represented a concern where specific conductivity exceeded 5 mmhos/cm. This could be attributed to chronic chloride toxicity which for all flow conditions averaged 152% of the 230 mg/L aquatic chronic criteria at station TS11 (USEPA, 1999).

This study extended previous work with *C. fluminea* to show that reduced growth and survivorship of the organism can be observed throughout a watershed impacted by AMD and commingled urban and agricultural land uses, on both mainstem and tributary reaches. Watershed ecological risk assessment of mining activities commingled with other land uses was successfully conducted using acute testing with *C. dubia* and chronic testing with *C. fluminea in situ*, together with water quality sampling. Attributed to unmonitored pollutants, mixed results were obtained in areas heavily impacted by urban or agricultural activities where Zn and Cu toxicity and upstream mining activity were absent. Consideration of drainage area in watershed-scale and subwatershed-scale ecological risk assessments will be required if *C. fluminea* growth rates are to be used to differentiate among various catchment sizes impacted by water quality degradation.

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Chapter 9 - ENGINEERING SIGNIFICANCE

Justin Eric Babendreier

Watershed improvement programs often attempt to assess and then change or control landscape patterns to benefit ecology, sediment and water quality, and overall stream health. This work focused on landscape influences associated with past and present coal mining operations. Six papers touched on common themes including watershed management, hydrologic study, and ecological risk assessment. Each manuscript attempted to advance our understanding of AMD impacts to ecology, integrating hydrologic data within multidisciplinary diagnostic programs. This was done on watershed and subwatershed scales, assessing stress associated with urbanization in small watersheds, agriculture, and various coal mining activities.

A term used here, hydroecological risk assessment, highlights the importance that both hydrology and ecology play in watershed management schemes. Within Civil and Environmental Engineering and other disciplines, it is likely that hydroecological-based engineering approaches will continue to grow in importance. Hydroecological-based watershed modeling applications may rise in response to these trends. Approaches will eventually be guided by the demands and monetary values placed on the quality and quantity of water and sediment draining watersheds. It is natural to expect that cost and benefit analysis of ecosystem services will ensue. Engineers will be called upon more frequently to place the value of ecology in some quantifiable perspective for land managers. Hydroecological approaches along lines developed here may one day be successful in reliably relating chemical, physical, and land use/cover variables directly to integrative biocriteria like IBI and ICI scores or other biometrics.

9.1 SIGNIFICANCE OF AUML LAND USES

Surface area of abandoned underground mines (AUML) above stream grade elevations was found to be strongly related to the occurrence and severity of untreated acid mine drainage (AMD). Aereal extents of strip mines (ASML) associated with the same coal bed though were not found to be good indicators of AMD or AMD impact. Increasing acidity and toxicity of AMD derived from AUML resulted in greater devastation observed in aquatic communities. This was also observed directly through ecotoxicological testing. AMD can be quantitatively related to functions of ecology, for example through water quality criteria for zinc during flow regimes describing drought, baseflow, run-off, and flooding conditions in a watershed. Linking these concepts in assessment design, one can begin to establish a concrete, technically sound hydrologic-hydraulic framework for attaining successful landscape management of abandoned mine lands (AML). Historically intractable problems of AMD carry severe long-term consequences to ecology and watershed economies if they are not efficiently addressed using scarce resources typically available for remedies.

Clean water and diverse ecology both can be viewed as scarce resources and each can be valued for different reasons. Metrics of clean water and diverse ecology will almost always be positively correlated. Engineering, biological, and ecological methods can inform each other of dominant landscape patterns leading to an improved understanding of watershed systems. An example of benefits to be gained by embracing multimedia and multidisciplinary approaches in watershed ecological risk assessment was shown here. The roles of hydrology and hydraulics were highlighted. Substantial benefits appeared to be realized by managing AML from an increasingly cost effective and technically sound finer-scale GIS approach. By quantifying and distilling the hydrologic landscape, land use/cover and flow regime considerations were important to understanding dominant patterns in aquatic biodiversity. Assessment approaches like %AUML might one day become significant and sufficient vantagepoints for ecological risk assessment.

9.2 INTEGRATION OF ECOLOGICAL RISK ASSESSMENT FOCUSES

Three basic components that drive aquatic response in environments with commingled urban, agricultural, and mining land uses are habitat, chemistry, and toxicity. A post-analysis overview of the ecological risk assessment paradigm presented in Figure 1.1 in Chapter 1 is summarized here with key findings. Emphasis was placed on key findings and mechanisms identified for Leading Creek, a 388 km^2 watershed in southeastern Ohio. In Leading Creek coal and limestone has been historically mined where two active coal mines are still in operation today, producing approximately 5.5 to 6 million tons of coal annually.

The concept of risk assessment was applied here in an ecological setting (USEPA, 1992, 1998). Ecological risk assessment was earlier defined as a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. Research presented here exploited a multidisciplinary, multimedia framework for conducting ecological risk assessment across watersheds. This was viewed from the standpoint of defining anthropogenic stressors using variables that describe land use/cover characteristics. The hydrologic-hydraulic based approach allowed enhanced assessment of landscape influences upon aquatic ecology from a perspective of subwatershed and watershed scales.

Condition of aquatic ecology as a response to stressors was first evaluated through analysis of habitat quality and insect biodiversity. A second focus was then investigated assessing signatures of AMD in both the sediment and water columns. Supporting these two approaches, a final, third focus was placed on examining ecotoxicity across water and sediment columns instream. Seen in Figure 1.1, production of materials from the landscape reaching channels is comprised of two phases, water and sediment. In this research, analysis of hydrologically driven responses instream were examined along this natural partition.

9.2.1 Landscape-Habitat-Biodiversity Relationships

In the first focus relating landscape variables to habitat and biodiversity, three basic questions were investigated: 1) What habitat parameters impacted aquatic biodiversity? 2) What dominant land use/cover groups influenced these key habitat scores? and 3) What dominant land use/cover groups impacted biodiversity?. These questions were evaluated using a combination of statistical techniques including ANOVA, correlation analysis, principal components analysis, and multivariate regression. Two separate qualitative habitat assessment procedures were used. Ohio EPA's QHEI or Qualitative Habitat Evaluation Index was found to be more significantly associated with land use/cover variables studied here than was USEPA's Rapid Bioassessment Protocol. It was determined that ASML>3% and Urban>5% impacted total QHEI habitat data, but active coal mining, minor ASML, and agriculture represented relatively insignificant stresses to habitat. For ASML>3%, with or without mixed AUML>2%, differences in habitat were attributed to increased sand fractions where QHEI pool-riffle subscores were also impacted. Attributed to siltation, Urban>3% to 5% impacted total QHEI and substrate subscores. Looking at relations between landscape and biodiversity, it was found that AUML>2%, Urban>5%, and to a lessor degree ASML>3% imparted significant stresses to biodiversity.

Of interest to evaluating the success of the assessment framework developed here, examination of insect biometrics allowed successful investigation and comparison among small and large subwatersheds (1 km^2 to 1000 km^2). Total and EPT insect taxon richness were more discriminating biometrics among coal mining land uses than were their counterpart abundance scores. On the other hand, EPT data showed significant sensitivity to land use/cover descriptions for small urbanized subsheds. The two variables taxon richness and abundance were generally shown to be closely linked by a power function expression (Taxa = Abundance^b) with root b = ½ for the sensitive EPT subgroup and b = 2/3rds for all insects, respectively. In examining landscape-habitat-biodiversity relationships at a scale of 1:24000, directly developed geodata from paper maps appeared to be more reliable than processed Landsat5 images in characterizing coal mining land uses and their impacts to habitat and biodiversity.

Overall, stress from AUML>2% was attributable to AMD where comparatively less stress to biota was associated with ASML>3%. Increasing %ASML in a given subshed appeared to degrade habitat through stream sedimentation and changes to morphological structure. Urbanization was associated with degraded habitat and particularly poor substrate scores.

9.2.2 AMD Signatures in Water and Sediments

Providing ability to signature chemicals from stressors to aquatic ecology offers a desirable capacity to connect landscape's cause directly to a toxicant effect instream. This work looked at coal-bed and sediment quality first, comparing data to variation in the dominant land use/cover in each subshed monitored. Origin and fate of Fe, Zn, Mn, and S in sediments was juxtaposed in a discussion of silt/clay versus sand size particle ranges in sediment. Examined were water column relations between pH, Zn, and Mn and soluble-solid dynamics in among TSS, Fe, Zn, and Mn data. Aside from identifying AUML as the most important variable in the expression of AMD, untreated AMD was well described *in situ* by its associated low pH and concomitant rise in zinc and sulfate concentrations. Mining signatures using Mg, Zn, and sulfate were found to be successful discriminators of AMD expression instream. In this approach, a two factor analysis was provided examining variation in AUML (<2%, 2% to 10%, 10%), and variation in flow regime described by drought, low, medium, high flow conditions. This approach allowed for examination of water and sediment chemical databases, accounting for variation in geology, land use/cover, and flow regime.

The geology of Leading Creek AML (= ASML+AUML) occurring above stream grade elevations was described by on overlying massive cap rock of sandstone, resting atop a 1m layer of shale/clay overlying the Pennsylvanian aged Redstone coal bed. A major finding for Leading Creek was that water quality, and not sediment quality, dominated risks to aquatic ecology derived from various coal mining activities. This was true for sites with water column pH as low as 3 to 4 s.u.. Sediment concentration values for Fe, Mn, Zn, and Cu across the watershed were all found to be within average levels described for earth's crustal abundance. Zn and Fe were observed to be correlated to solids concentrations associated with silt/clay (washload). Though some discrimination of land use/cover was observed in individual concentrations of Fe and Zn, the logZn/logFe ratio was relatively indifferent to variation in land uses/cover, with some minor exceptions for ASML>3%. Consistent Fe and Zn solid-soluble dynamics across land uses were attributable to mechanisms of instream precipitation and sedimentation. Zn and Fe were associated with pyretic shaly/clay layers surrounding coal seams, and increases in Fe and Zn in washload increased during high flow conditions, indicating suspension of Fe and Zn rich solids.

Mn on the other hand was attributed to sandstone materials and sand fractions, and was not associated with significant mechanisms of instream precipitation and sedimentation. With a paucity of Mn seen in and around shaly/clay and coal bed seams, and a paucity of Mn seen in some acidified streams, Mn appeared to be scoured into the water column from coarser sandstone sediments and soil. For ASML land uses >3% by area, a higher relative fraction of Mn and to a lessor degree Zn were found in sediments compared to Fe. Water column Mn tracked Zn except in drought where Mn levels were observed to rise in subsheds without significant stream acidification or mining activities. This was attributed in part to evaporation and lack of mechanisms sinking Mn to sediments due to slower rates of oxidation in natural environments.

Low pH, and elevated Zn and SO_4 characterized untreated AMD which derived primarily from AUML>2% and was worst at baseflow. Water column Mg/SO4 ratios also were found to uniquely identify treated AMD apart from untreated AMD, and could differentiate AMD apart from limestone quarry discharges instream. A critical aspect of AMD signatures was their ability to provide a separate line of evidence imparting cause and effect between geology, land use/cover, hydraulics, chemistry, and decreased biodiversity. The focus on AMD signatures for example helped identify the sources of AMD metals appearing instream, relating these levels back to coal bed and soil quality. Transport and fate of AMD contaminants downstream was found to be consistent across examinations of water and sediment columns. Zinc and pH coupled with assessment of flow condition were found to be the most promising parameters with respect to their ability to tie lower biodiversity scores to AMD impacts which were tied to AUML land uses.

9.2.3 Ecotoxicological Assessment of Water and Sediments

Ecotoxicity was studied to support two prior focuses discussed including examination of landscape-habitat-biodiversity relations and determining AMD signatures in the sediment and water columns. Ecotoxicity provided a third line of evidence and supported the assertion that the existence of AUML derived toxicants instream were responsible for degraded biodiversity. Tying in ecotoxicity, the major finding between contaminant phases was again supported where water quality mechanisms and associated concerns were found to dominate conditions of increased risk to aquatic ecology. Water quality toxicant concerns were found for active and abandoned coal mining activities associated with underground mining. In general, increased zinc toxicity during increasingly lower flow conditions supported ecotoxicity observed for AUML>2%. For active underground mine operations discharging treated AMD, drier flow conditions and increased chlorides appeared to exhibit chronic toxicity responses in the Asian clam studied *in situ*.

Chemical analysis, ecotoxicological analysis, and land use cover analysis indicated that overall, ASML was not associated with water quality or sediment quality degradation, only habitat degradation. Degradation of biodiversity induced by toxicants was a function of water quality impact, not sediment quality. Acute and *in situ* tests reliably indicated AMD severity, corroborating land use/cover and chemical analyses. It was finally found that for subsheds in Leading Creek, *in situ* shell length growth rates of *C. fluminea* were significantly related to drainage area of the catchment, above and below a threshold area of 75 km^2 for some test events. Chronic *in situ* and acute stormwater run-off toxicity testing indicated moderate toxicity in a few urbanized and agriculturally disturbed subsheds, but responses were typically sporadic and inconsistent across multiple sampling events.

9.2.4 Simplified Conclusions

In summarizing important features of this work, it was found that (1) strip mining >3% by area moderately impacted biodiversity, degrading habitat quality; (2) AUML above stream grade elevations was the source of most AMD reaching Leading Creek and led to aquatic toxicity, impacting biodiversity when >2% by area, and devastating biodiversity when >10%; and (3) urbanization in small subsheds >5% by area affected both habitat and water quality. Similar to results found for a few agriculturally influenced sites, for urbanization poor water quality and some toxicity was sporadically observed. In light of flow regime considerations, treated and untreated AMD impacts in Leading Creek appeared to be more associated with baseflow conditions while instream impacts to water quality in urban and agriculturally dominated subsheds were more associated with run-off condition.

Appendix A - Land Use and Land Cover Data

Justin Eric Babendreier

Table A1 – Subsheds of Leading Creek. Monitored tributary and mainstem subsheds in downstream order; (Ref) – Reference, (TS) – Tributary, (LCS) – Mainstem. A total of 91 subsheds delineated in associated mapping represent monitored tributaries, tributary mouths below stations, unmonitored tributaries, and mainstem areas not drained by tributaries. A total of 61 primary tributaries, including the east and west headwater branches below Albany, Ohio confluence the mainstem of Leading Creek (scale 1:24000).

Station ID	Station Name	Station Code	Latitude	Longitude	Manual Staff Gage Station ID	Continuous Gage Station ID	River Kilometer (km)	Distance to Ohio River (km)	Drainage Area (ha)	Stream Order
(R)FCS1	Upper Federal Creek	REF	39.415283	-81.961336	-	-	-	-	6789	-
(R)SCS1	Upper Symmes Creek	REF	38.787715	-82.428035	-	-	-	-	34818	-
(R)SCS2	Lower Symmes Creek	REF	38.477033	-82.440042	-	-	-	-	89637	-
TS1	West Fork Headwater	TS	39.216994	-82.218339	-	-	53.07	53.07	124	2
TS2	East Fork Headwater	TS	39.220244	-82.215183	-	-	0.50	53.29	169	2
LCS1	Albany	LCS	39.203489	-82.201706	SGS1	-	50.66	50.66	998	3
TS3A	Five Mile Run	TS	39.165908	-82.207079	SGS2	-	1.48	45.78	721	3
TS3	Five Mile Run	TS	39.164408	-82.209544	-	-	1.15	45.46	1126	2
LCS2	Carpenter	LCS	39.161131	-82.222280	SGS3	-	44.16	44.16	3447	4
TS4	Sharps Run	TS	39.162165	-82.231441	-	-	1.15	44.70	969	3
LCS3	Sisson	LCS	39.142113	-82.234614	SGS4	-	41.06	41.06	4829	4
TS5	Sisson Run	TS	39.143157	-82.240601	-	-	0.11	40.49	1446	3
TS6	Ogdin Run	TS	39.126853	-82.238862	-	-	0.43	37.49	1894	4
LCS4	Dyesville	LCS	39.121335	-82.229800	SGS5	-	36.21	36.21	8609	5
TS7	Dyesville Run	TS	39.121319	-82.223912	-	-	0.79	35.35	611	2
TS8A	Mud Fork	TS	39.132084	-82.184002	-	-	5.97	37.97	1846	3
TS8B	Mud Fork	TS	39.106211	-82.219802	SGS6	-	0.06	32.07	3412	3
TS9	Dexter Run	TS	39.095644	-82.231855	-	-	1.21	32.49	1897	4
TS10	Grass Run	TS	39.086436	-82.200424	-	-	1.33	29.70	482	2
LCS5A	Dexter North	LCS	39.083005	-82.214098	-	-	28.29	28.29	16085	5
LCS5B	Dexter South	LCS	39.073956	-82.209970	SGS7	CGS1	26.90	26.90	16229	5
TS11	Parker Run	TS	39.064782	-82.227255	SGS8	-	2.60	28.87	1476	3
LCS6	Malloons	LCS	39.062123	-82.200084	SGS9	-	24.87	24.87	18453	5
TS12	Malloons Creek	TS	39.062290	-82.203818	-	-	0.13	24.70	1072	3
LCS7	Corn Hollow	LCS	39.039592	-82.159066	SGS10	-	17.26	17.26	20918	5
TS13	Lasher Run	TS	39.028902	-82.148830	-	-	0.88	15.81	459	2
TS14	Little Leading Creek	TS	39.030611	-82.138550	SGS11	-	0.65	14.84	6467	4
TS15	Titus Run	TS	39.011095	-82.143205	-	-	0.34	12.67	693	2
LCS8	Twin Bridges	LCS	39.012779	-82.138167	SGS12	-	12.04	12.04	29243	5
TS16	Paulins Hill South	TS	39.002076	-82.131190	SGS13	-	0.89	11.16	157	1
LCS9	Iron Bridge	LCS	39.008742	-82.085342	SGS14	CGS2	5.82	5.82	30328	5
TS17	Thomas Fork	TS	39.004767	-82.074270	SGS15	-	2.00	4.61	7963	4
LCS10	Middleport	LCS	38.992435	-82.074534	SGS16	-	2.45	2.45	38662	5
OH	Ohio River	LCS	38.983700	-82.072385	-	-	0	0	38801	5

Table A2 – Abandoned Surface and Underground Mine Lands (AML). Areal extent of abandoned strip mines (ASML) and abandoned underground mines (AUML) shown as a percentage of subshed drainage area. Data includes reclaimed surface mines. Incorporates digitized data from U.S. Geological Survey (USGS) quadrangle maps and Ohio Division of Natural Resources (ODNR) abandoned underground mine inventories (scale 1:24000). Time periods shown for strips mines and reclaimed strip mines refer to actual mining dates, not dates of reclamation. Data excludes all historic and current Southern Ohio Coal Company (SOCCO) underground long-wall mine operations, including underground and above ground activities.

Station ID	Abandoned Strip Mines (ASML)			Reclaimed Strip Mines (RASML)			Abandoned Underground Mines (AUML)	Unreclaimed ASML + AUML
	<1961	1961 to 1975	Total	<1961	1961 to 1975	Total		
(R)FCS1	-	-	2.10%	-	-	0%	0.06%	2.15%
(R)SCS1	-	-	3.77%	-	-	0.49%	0.97%	4.25%
(R)SCS2	-	-	2.65%	-	-	0.30%	0.44%	2.80%
LCS1	0%	0%	0%	0%	0%	0%	0%	0%
LCS2	0%	0%	0%	0%	0%	0%	0%	0%
LCS3	0%	0%	0%	0%	0%	0%	0%	0%
LCS4	0%	0%	0%	0%	0%	0%	0%	0%
LCS5B	0.62%	0.35%	0.97%	0.16%	0.19%	0.35%	0%	0.62%
LCS6	0.55%	0.36%	0.90%	0.14%	0.17%	0.31%	0%	0.60%
LCS7	0.74%	0.37%	1.11%	0.14%	0.15%	0.29%	0%	0.82%
LCS8	2.35%	1.00%	3.35%	0.70%	0.32%	1.02%	0.53%	2.85%
LCS9	2.79%	0.99%	3.78%	0.67%	0.31%	0.99%	0.93%	3.72%
LCS10	3.57%	0.91%	4.48%	0.57%	0.26%	0.83%	4.16%	7.81%
TS1	0%	0%	0%	0%	0%	0%	0%	0%
TS2	0%	0%	0%	0%	0%	0%	0%	0%
TS3	0%	0%	0%	0%	0%	0%	0%	0%
TS4	0%	0%	0%	0%	0%	0%	0%	0%
TS5	0%	0%	0%	0%	0%	0%	0%	0%
TS6	0%	0%	0%	0%	0%	0%	0%	0%
TS7	0%	0%	0%	0%	0%	0%	0%	0%
TS8A	4.17%	1.37%	5.54%	0.46%	1.00%	1.46%	0%	4.08%
TS8B	2.75%	0.87%	3.62%	0.74%	0.67%	1.41%	0%	2.21%
TS9	0%	0%	0%	0%	0%	0%	0%	0%
TS10	0.92%	4.41%	5.33%	0.22%	1.66%	1.88%	0%	3.45%
TS11	0%	0%	0%	0%	0%	0%	0%	0%
TS12	0%	0%	0%	0%	0%	0%	0%	0%
TS13	9.35%	1.37%	10.72%	0%	0%	0%	0%	10.72%
TS14	5.62%	2.90%	8.53%	2.67%	0.98%	3.65%	1.67%	6.54%
TS15	12.17%	3.10%	15.27%	0%	0%	0%	2.79%	18.05%
TS16	30.33%	0%	30.33%	0%	0%	0%	19.53%	49.86%
TS17	6.08%	0.55%	6.63%	0.22%	0.07%	0.28%	15.86%	22.21%

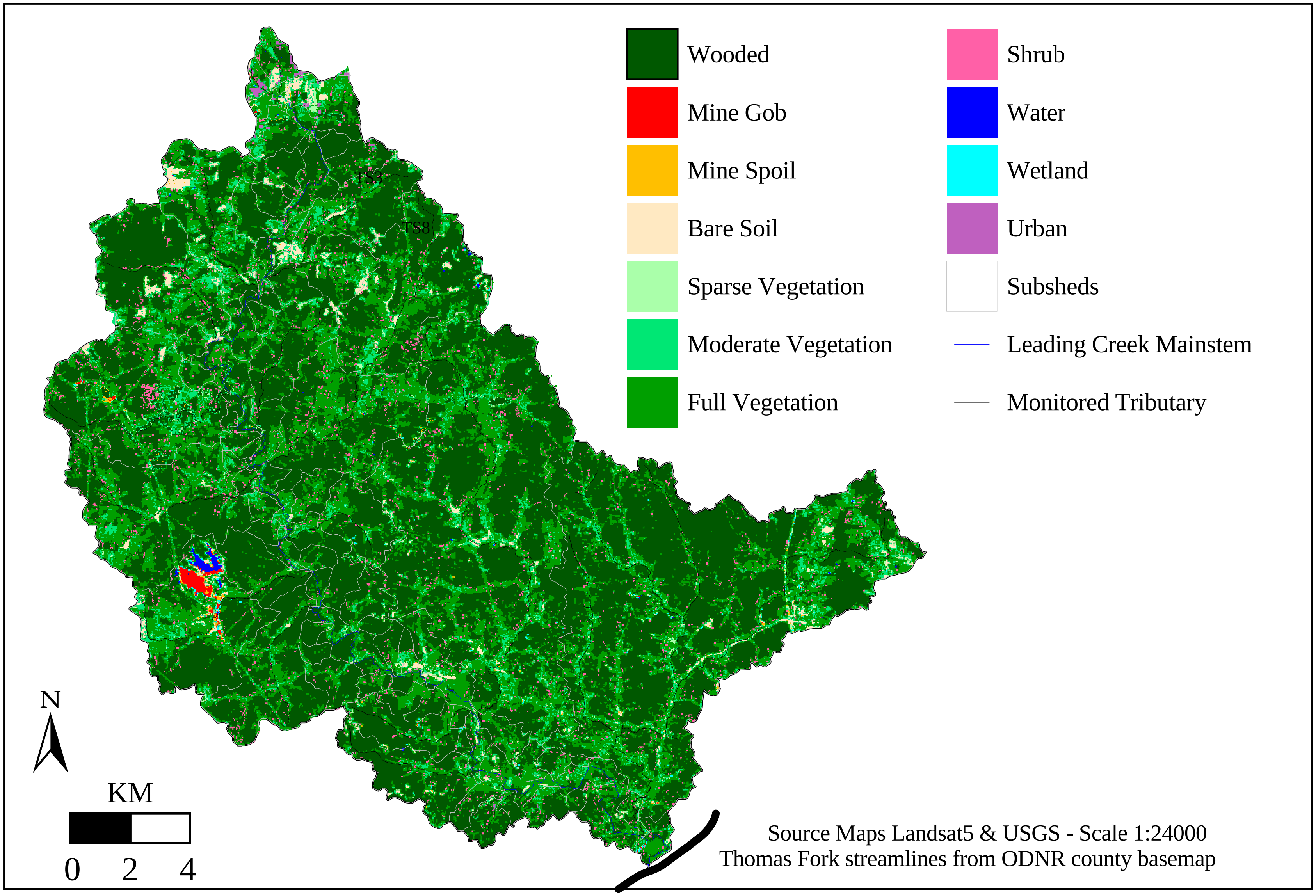

Figure A1 – June 1988 Landsat5 Land Use Cover Data. Landsat5 remotely sensed Thematic Mapper (TM) data interpreted by Ohio Department Natural Resources specifically to show the land cover of the coal mining region in Ohio.

Table A3 – 1988 Landsat5 Land Use Cover Data. Areal extent of Landsat5 remotely sensed Thematic Mapper (TM) data from June 1988; represented as a percentage of subshed drainage area. TM data interpreted by Ohio Department Natural Resources. Full TM coverage for Symmes and Federal Creeks was not readily available for this study. Active underground SOCCO mine operations in Leading Creek described as gob and spoil represent land surface coal product storage or processing areas, but are primarily waste material aggregate disposal areas. The large mine operation area in Parker Run delineated as water represents the reservoir polishing pond which stores treated AMD prior to discharge and is also used for process make-up water for plant operations.

Station	With SOCCO Mines		Without SOCCO Mines				Vegetation						
ID	% Mine Gob	% Mine Spoil	% Mine Gob	% Mine Spoil	%Urban	%Bare Soil	% Sparse	% Moderate	% Full	%Shrub	%Wooded	%Water	%Wetland
LCS1	0.12%	0.37%	0.12%	0.37%	4.02%	8.26%	2.71%	13.17%	38.67%	2.25%	30.22%	0.05%	0.16%
LCS2	0.04%	0.13%	0.04%	0.13%	1.24%	4.42%	1.31%	8.82%	29.07%	2.20%	52.69%	0.02%	0.06%
LCS3	0.03%	0.14%	0.03%	0.14%	0.89%	4.45%	1.23%	9.26%	30.96%	2.21%	50.75%	0.02%	0.07%
LCS4	0.06%	0.16%	0.02%	0.08%	0.50%	4.62%	1.15%	8.84%	29.82%	2.73%	52.05%	0.01%	0.06%
LCS5B	0.04%	0.11%	0.01%	0.07%	0.26%	2.94%	0.84%	7.34%	31.08%	2.52%	54.76%	0.03%	0.07%
LCS6	0.44%	0.22%	0.01%	0.06%	0.23%	2.78%	0.84%	7.05%	30.26%	2.40%	55.37%	0.24%	0.15%
LCS7	0.39%	0.20%	0.01%	0.06%	0.20%	2.54%	0.80%	6.82%	29.55%	2.34%	56.80%	0.21%	0.14%
LCS8	0.28%	0.17%	0.01%	0.07%	0.15%	2.15%	0.79%	6.61%	29.32%	2.19%	58.06%	0.16%	0.13%
LCS9	0.27%	0.16%	0.01%	0.07%	0.15%	2.12%	0.80%	6.64%	29.67%	2.19%	57.70%	0.15%	0.13%
LCS10	0.22%	0.16%	0.01%	0.08%	0.12%	1.88%	0.81%	6.54%	28.79%	2.06%	59.16%	0.13%	0.13%
TS1	0.95%	0.62%	0.95%	0.62%	12.69%	7.87%	2.22%	15.89%	50.99%	2.22%	5.82%	0.15%	0.58%
TS2	0%	0.05%	0%	0.05%	7.01%	6.69%	1.13%	11.95%	38.38%	4.31%	30.31%	0.16%	0%
TS3	0.01%	0.02%	0.01%	0.02%	0%	3.85%	1.05%	9.13%	30.00%	2.27%	53.67%	0%	0.01%
TS4	0.01%	0.25%	0.01%	0.25%	0%	5.81%	1.19%	11.64%	36.60%	2.21%	42.10%	0.07%	0.12%
TS5	0%	0.01%	0%	0.01%	0%	2.93%	0.82%	8.53%	23.52%	2.78%	61.39%	0%	0.02%
TS6	0.19%	0.33%	-	-	0%	6.36%	1.26%	8.73%	30.76%	3.89%	48.37%	0%	0.10%
TS7	0%	0.01%	0%	0.01%	0%	0.58%	0.49%	4.19%	25.28%	1.89%	67.56%	0%	0%
TS8A	0.03%	0.06%	0.03%	0.06%	0%	2.34%	0.98%	6.34%	29.57%	2.30%	58.12%	0.13%	0.12%
TS8B	0.03%	0.09%	0.03%	0.09%	0%	1.46%	0.66%	5.45%	33.84%	2.48%	55.80%	0.11%	0.09%
TS9	0%	0.03%	0%	0.03%	0%	1.09%	0.34%	5.83%	35.31%	2.43%	54.93%	0%	0.03%
TS10	0%	0.05%	0%	0.05%	0%	0.10%	0.56%	5.50%	25.51%	1.28%	66.63%	0%	0.37%
TS11	5.15%	1.59%	-	-	0%	2.27%	1.20%	7.02%	29.25%	1.75%	47.97%	2.68%	1.12%
TS12	0%	0%	0%	0%	0%	0.22%	0.25%	4.48%	25.29%	2.02%	67.74%	0%	0%
TS13	0.07%	0.29%	0.07%	0.29%	0%	0.64%	0.73%	3.30%	22.96%	1.83%	69.79%	0.06%	0.34%
TS14	0.01%	0.07%	0.01%	0.07%	0%	1.05%	0.73%	6.29%	28.42%	1.85%	61.49%	0.02%	0.09%
TS15	0%	0.19%	0%	0.19%	0%	1.33%	0.42%	5.07%	20.05%	1.42%	71.39%	0.05%	0.08%
TS16	0%	0.06%	0%	0.06%	1.33%	0.18%	0.52%	2.21%	20.98%	1.91%	72.82%	0%	0%
TS17	0.02%	0.12%	0.02%	0.12%	0%	1.00%	0.81%	6.15%	24.99%	1.50%	65.20%	0.03%	0.17%

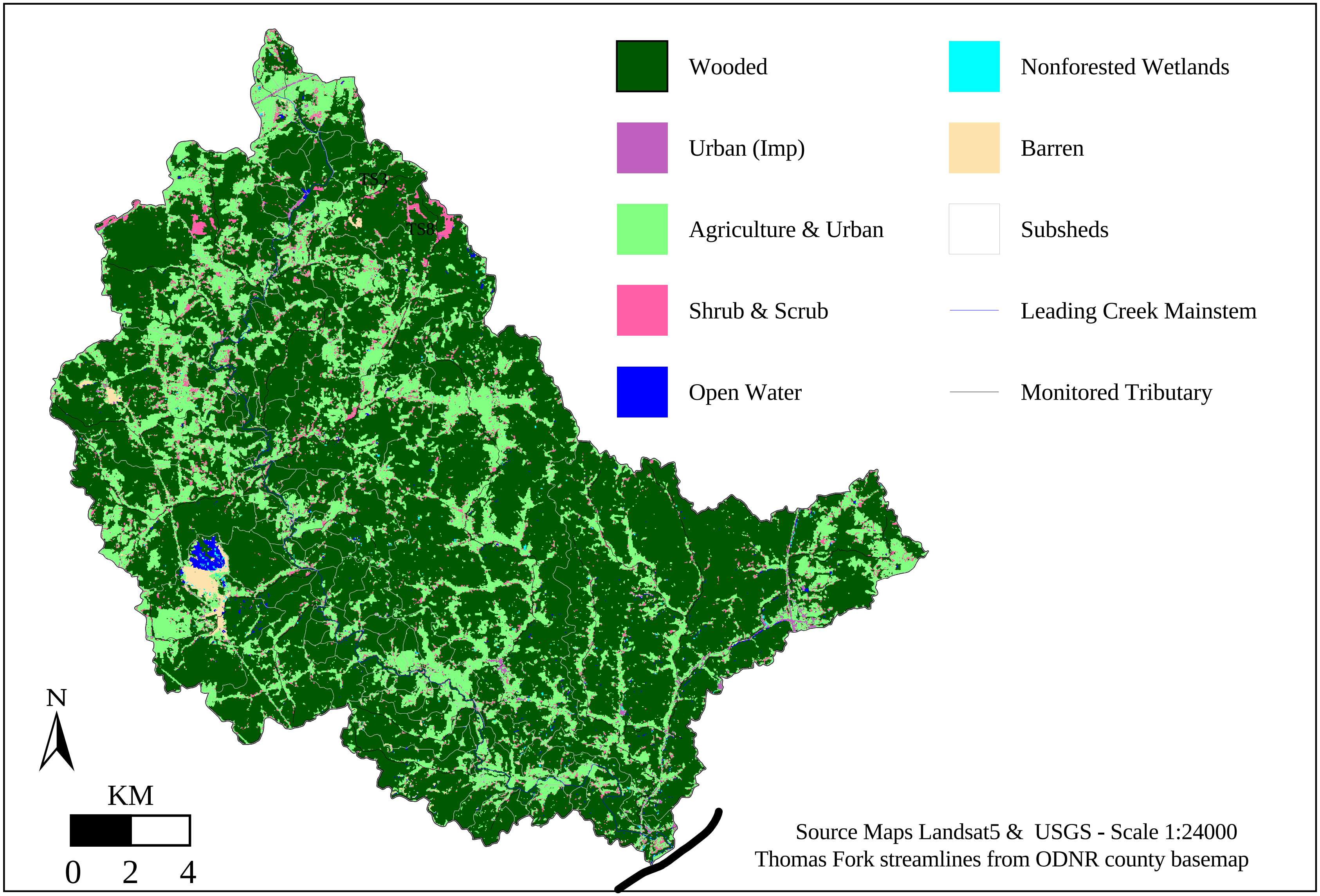

Figure A2 – September and October 1994 Landsat5 Land Use Cover Data. Landsat5 remotely sensed Thematic Mapper (TM) data interpreted by Ohio Department Natural Resources specifically for NASA'a Land Cover and Land Use Change Program.

Table A4 – 1994 Landsat5 Land Use Cover Data. Areal extent of Landsat5 remotely sensed Thematic Mapper (T M) data from September and October 1994; represented as a percentage of subshed drainage area. TM data interpreted by Ohio Department of Natural Resources (ODNR) for the National Aeronautics and Space Administration's (NASA) Land Cover and Land Use Change Program as part of their U.S. Global Change Research Program. According to ODNR, the data was intended for use in research programs dealing with flooding, erosion, recreation, water resources, land use development, habitat, wetlands, natural areas, and fish and wildlife. The 1994 Landsat5 data interpretation performed by ODNR incorporated the State of Ohio's 1986 Ohio Wetlands Inventory coverages.

Station ID	%Impervious Urban	%Agriculture & Open Urban	%Shrub & Scrub	%Wooded	% Open Water	%Non-Forested Wetland	%Barren
(R)FCS1	0.43%	27.12%	4.15%	68.10%	0.08%	0.08%	0.04%
(R)SCS1	1.35%	37.52%	3.16%	55.61%	0.46%	0.61%	1.30%
(R)SCS2	0.81%	28.76%	3.02%	66.23%	0.36%	0.27%	0.55%
LCS1	2.03%	47.17%	5.55%	44.11%	0.41%	0.21%	0.52%
LCS2	0.95%	27.55%	4.71%	65.98%	0.29%	0.09%	0.42%
LCS3	0.77%	29.73%	5.13%	63.68%	0.24%	0.09%	0.36%
LCS4	0.59%	28.59%	5.01%	65.03%	0.18%	0.08%	0.52%
LCS5B	0.44%	26.08%	4.60%	68.26%	0.17%	0.10%	0.34%
LCS6	0.48%	25.17%	4.31%	68.20%	0.61%	0.16%	1.06%
LCS7	0.48%	24.48%	4.05%	69.34%	0.55%	0.15%	0.95%
LCS8	0.51%	23.80%	3.57%	70.77%	0.43%	0.16%	0.76%
LCS9	0.50%	23.83%	3.55%	70.79%	0.43%	0.16%	0.75%
LCS10	0.68%	22.38%	3.30%	72.48%	0.40%	0.17%	0.61%
TS1	6.29%	81.14%	3.26%	6.72%	0.40%	1.16%	1.03%
TS2	0.27%	40.18%	9.46%	49.39%	0.37%	0.33%	0%
TS3	0.19%	25.44%	5.43%	68.06%	0.04%	0.03%	0.80%
TS4	0.21%	36.78%	7.19%	55.62%	0.05%	0.06%	0.08%
TS5	0.29%	23.34%	4.64%	71.58%	0.06%	0.05%	0.04%
TS6	0.45%	29.94%	4.78%	63.23%	0.13%	0.09%	1.38%
TS7	0.14%	15.03%	2.61%	82.14%	0.00%	0.07%	0.01%
TS8A	0.29%	21.35%	5.46%	72.39%	0.28%	0.15%	0.08%
TS8B	0.25%	23.94%	5.11%	70.23%	0.24%	0.12%	0.11%
TS9	0.34%	26.35%	3.83%	69.12%	0.06%	0.05%	0.24%
TS10	0.11%	17.99%	2.90%	78.19%	0.26%	0.38%	0.17%
TS11	1.06%	24.59%	2.23%	56.00%	5.63%	1.00%	9.49%
TS12	0.42%	17.42%	2.00%	79.98%	0.06%	0.06%	0.05%
TS13	0.39%	17.87%	1.24%	79.43%	0.08%	0.14%	0.84%
TS14	0.56%	22.10%	2.50%	74.41%	0.13%	0.18%	0.13%
TS15	0.60%	13.18%	1.24%	82.95%	0.39%	0.08%	1.56%
TS16	0%	11.01%	1.61%	87.07%	0.04%	0.00%	0.28%
TS17	1.28%	17.23%	2.33%	78.57%	0.30%	0.19%	0.10%

Land Use Cover Data Definitions and Development

Spatial analysis of subsheds used several sources including data hand-digitized from USGS quadrangle 7.5-minute map series (scale 1:24000), field data, and USGS hydrology geodata obtained from Ohio Division of Natural Resources (ODNR). Land use/cover data (scale 1:24000) was developed through four primary sources including a) 7.5-minute quadrangle series maps depicting abandoned strip mines, b) Ohio Division of Natural Resource (ODNR) geodata depicting abandoned underground mines, c) 1988 Landsat5 Thematic Mapper (TM) geodata developed by ODNR specifically for assessment of abandoned mine lands, and d) 1994 Landsat5 TM geodata developed by ODNR for general environmental assessment. Multi-spectral TM processing evaluates visible and infrared electromagnetic radiation from the earth's surface.

Leading Creek subsheds were defined by digitizing hand drawn rain fall lines that also incorporated interpretation of the major influences of active drainage management associated with roadways. Initial conversion and processing of basemap ARC/INFO® exchange file formats for ODNR geodata was carried out using either using ArcView® 3.0 and later versions or the ATLAS® Import/Export program. USGS strip mine data was maintained for two time periods (pre-1961, and 1961 to 1975) and % by surface area of each subshed covered by strip mines was determined. Coverages for reclaimed ASML were also developed. Thomas Fork, Federal Creek and Symmes Creek strip mines were calculated by hand using USGS quadrangles (scale 1:24000) and grid counting (1 unit = 0.1062 ha) with precision recovery < 5%.

The outer watershed boundary for Leading Creek was later reconciled with USGS HUC hydrology coverage provided by ODNR (scale 1:24000) from which basins for Federal Creek, and Symmes Creek were also developed. Actual subsheds monitored in the reference watersheds were also cut based on fall analysis. Final calculations for all subsheds reconciled several sources of spatial data including the rubbersheeted 7.5-minute USGS quadrangle data (scale 1:24000) digitized by Virginia Tech and rubbersheeted statewide underground mine coverages (scale 1:24000) provided by ODNR. Virginia Tech digitization was conducted using a 12"x12" Summa Sketch digitizer board and employed a minimum of 7 control points with maximum acceptable control point digitization error of 0.005 map units. An extensive quality assurance check was completed on all portions digitized by Virginia Tech where map resolution for cleaning and feature vertex checks were examined at scales of ~ 1:200-500. Final coverages were all inspected to assure mutually exclusive and complete coverage of subject watersheds.

Data describing locations of study sites, stream order, drainage area, stream length, and associated land use/cover characteristics were calculated using ATLAS-GIS® Version 3.01 for Windows® and linked Microsoft Office® 97 spreadsheet-database programs. All statistical analysis was performed using the NCSS® 97 software package (Hintze, 1997) or routines in the Excel spreadsheet package. All raw and processed data was managed via linked ACCESS and DBASE III databases and EXCEL pivot tables. Streams except Thomas Fork were digitized from USGS 7.5-minute series quadrangles using hand digitization and other GIS techniques. Three hundred two separate stream segments (Strahler, 1957) excluding Thomas Fork's tributaries comprise 61 primary tributaries that directly confluence the mainstem of Leading Creek. The range of drainage areas describing the 61 primary tributaries and associated subsheds identified in Figures A1 and A2 span approximately 37 to 8038 ha each. Table A1 gives specific drainage areas for each routinely monitored subshed and quantified geographic locations of cross-sections studied. Thomas Fork streamlines were derived from the Meigs County digital line graph basemap available from ODNR. Two complete data sets of Landsat5 Thematic Mapper data (resolution = 30x30m) describing watershed land use and land cover characteristics were incorporated. Described in Tables A.5 and A.6, definitions, processing and interpretation were provided by ODNR.

Table A5 - June 1988 Landsat5 Thematic Mapper Data (ODNR)

Layer	GRID-CODE	Category Description
1	Mine Gob	Darkest coal wastes
2	Mine Spoil	Coal waste mixed with subsoil and topsoil
3	Bare Soil	
4	Sparse Vegetation	Vegetation cover less than 50%
5	Moderate Vegetation	Vegetation cover between 50% and 100%
6	Full Vegetation	100% vegetation cover (mostly herbaceous)
7	Shrub	
8	Wooded	
9	Water	
10	Wetland	
11	Urban	Urban, residential, and commercial

All GIS data and area calculations were standardized to geographic coordinates using the North American Datum (NAD) of 1927 and retain positional accuracy ± 15m. All ODNR land use-cover data processed from Landsat5 was generated as non-topologic vector data processed originally as raster data using ERDAS software and referenced to Universal Transverse Mercator zone 17 coordinates based on NAD 1927. USGS, ODNR, USDA and Landsat5 coverage data were not typically field checked, but were usually well reconciled with extensive field visits and observations conducted between 1995 and 1998. Bare soil data in Table A.5 was interpreted here to represent disturbed agricultural lands at spring planting and heavy livestock operations, with some influences attributable to urban features like road shoulders. Only portions of the 1988 Landsat5 data coverage were available for Symmes Creek and Federal Creek reference watersheds and therefore could not be evaluated. Land use/cover data for 1994 represents less defined segregation of urban and agriculture and was developed from remote sensing in fall of that year. The 1994 ODNR coverage also incorporates 1986 Ohio Wetlands Inventory geodata.

Table A6 - September 1994 Landsat5 Thematic Mapper Data (ODNR)

Layer	GRID-CODE	Category Description
1	Urban	Open impervious surfaces: roads, buildings, parking lots and similar hard surface areas that are not obstructed from areal view by tree cover. See BARREN
2	Agriculture & Open Urban Areas	Cropland and pasture; parks, golf courses, lawns and similar grassy areas not obstructed from view by tree cover (referenced here as Ag/Urban).
3	Shrub/Scrub	Young, sparse, woody vegetation; typically areas of scattered young tree saplings.
4	Wooded	Deciduous and coniferous.
5	Open Water	
6	Non-Forested Wetlands	Includes wetlands identified from 1994 Thematic Mapper data as well as from the 1986 Ohio Wetlands Inventory.
7	Barren	Strip mines, quarries, sand and gravel pits, and beaches. Many of the URBAN features identified in this inventory are constructed from materials obtained from the BARREN features. Because of this, there will on occasion be URBAN areas identified as BARREN as well as BARREN areas identified as URBAN.

Remotely Sensed Land Use and Land Cover Data

Assignment of Bare Soil as heavy agricultural activity was consistent with recent field surveys for all subsheds except headwaters stations TS1 and TS2 which drain portions of the Town of Albany, Ohio. Small subsheds are more subject to changes of land use over time. According to the town water clerk, the large disturbance noted in 1988 due north of State Route 12 and Dickson Road was part of a quarry operation in the area at the time. During 1995 to 1998 data collection the quarry had moved due northwest and was no longer active at those sites, replaced this decade by low-density housing construction activity. The local golf course manager confirmed that the quarry was the only significant disturbance in those subsheds in 1988, supporting a dominant urban land use designated here for stations TS1 and TS2.

Due to geo-referencing errors in ODNR 1994 Landsat5 land use/cover data for Lawrence County, associated geodata had to be proportionately scaled to ATLAS county line data. ATLAS county line data was of poorer quality. Land use/cover estimates for Lower Symmes Creek station (R)SCS2 have additional error potential of up to 12% to 15% based on scaling differences observed with well referenced data. Only 3.5% of Upper Symmes Creek station (R)SCS1 drainage area fell within Lawrence County.

Landsat5 gob and spoil identified near active underground mining areas in Figure 3.4 represented waste rock and coal processing operations. Mine gob and spoil identified in tributaries believed to be not mined indicated that remotely sensed mining residuals may have been difficult to distinguish from natural clastic materials not associated with coal mining. An example of this was TS1 where significant portions of an airport operation were identified as mine gob. Mine spoils were more frequently identified in subsheds not mined than was mine gob. Analysis here incorporated classification of active mine operations with and without gob and spoil. USGS 1:24000 maps based on 1991-1992 photography incorrectly classified gob and spoil piles at SOCCO's active underground mines as strip mined areas, the same large areas identified as Barren in Figure 3.4. The active underground coal mine operations result in surface disposal of waste material (i.e. clastic aggregate) where stormwater runoff is actively managed by SOCCO using a 10-year design basis. Recent mining activities manifested at the surface were apparently difficult to discern from strip mines based solely on analysis of aerial photography.

Directly Sensed Land Use and Land Cover Data

Digitized USGS quadrangle data in Leading Creek representing strip mines was based on maps available up to 1995. Revised USGS maps made available after 1995 were not directly incorporated, representing primarily only minor changes in strip mining designations associated with waste pile activities from active underground mines. USGS quadrangle mapping for Federal Creek and Symmes Creek strip mining designations were based on USGS quadrangle maps available up to years 1999 and 2000, respectively. Few reclaimed strip mines were noted to date in the Thomas Fork tributary (TS17) and none were noted in Leading Creek tributaries west of the mainstem. Thomas Fork data is not shown in Figure 3.6 but occurrence of strip mines is well described in this subshed by often co-located abandoned underground mines seen in Figure 3.7. No distinction was made between strip mining time periods in the reference watersheds Symmes Creek and Federal Creek. Based on aerial photography updates, designation of 1961 and 1975 revision dates may be slightly different in Leading Creek depending on each USGS quadrangle. Federal Creek data was believed to be associated with pre-1961 mining activities, and Symmes Creek was a mixture of both periods.

For Symmes Creek, status of reclamation was based on the most recent USGS 7.5-minute quadrangle mapping notations available in 2000. No indications of ASML reclamation status were noted by USGS on Leading Creek or Federal Creek quadrangle maps. Primary determination of reclamation status of strip mines in Leading Creek was taken from the designated soil series Pinegrove Course Sandy Loam (Pn) and Pinegrove Silty Clay Roam (Pu) representing unreclaimed and reclaimed lands, respectively (Gillmore et al, 1991). Leading Creek soil survey maps and data were supplemented by metadata from the Abandoned Mine Land Inventory System (AMLIS) database and mapping provided by the local USDA Soil Conservation Service field office. AMLIS reclamation designations do not typically incorporate strip mine areas that have been partially reclaimed. The USDA soil survey designation was the most methodically developed description of land surface and cover characteristics associated with strip mining in Leading Creek. The (Pu) and (Pn) soil series incorporate activities of reclamation at the time of mining and included later interventions resulting in larger areas identified with soil and vegetative cover.

In Leading Creek soil surveys did not always coincide with AML program data, descriptions, and agency information that otherwise indicated most strip mine areas had been reclaimed by 1998. Differences were evident in use of the term reclamation. County soil data referred to surface condition status, and other definitions by USGS and USDA implied activity that may or may not have been successful, and disturbed land inventories that may not have been complete. Compared to other data sources, soil survey data classifying surface mining activity appeared to be a more comprehensive inventory and is inherently quantitative. Based on aerial photography at scale 1:15840, only non-digital soil survey data was available in Leading Creek. With respect to size, shape, and location, good agreement was found between USDA soil survey (Pu) and (Pn) designations and digitized USGS strip mine classification (scale 1:24000), based on visual comparisons. In only a few cases were strip mine areas identified in one data set but not the other. In one case AMLIS reclamation designation was used to supplement a county soil series designation, changing (Pn) to (Pu). AMLIS mapping was assumed to be more current.

Leading Creek coverage of abandoned underground mines in Figure 3.7 represents 124 non-overlapping coal mine works where all reside above stream drainage elevations. Upper Symmes Creek includes some overlaying mines with multiple extraction products including clay and some limestone. As an example, these latter operations might include a base floor of clay also extracted. Long wall mining operation aereal extents shown (>100m BLS) at Mines Nos. 2 and 31 cover 6300 ha and represent deep groundwater diversion zones associated with treated AMD discharged into Leading Creek. The most western section of active mine No. 31 shown in Figure 3.7 is closed and currently flooded. This is the zone where an in place bulkhead seal failed, previously separating the active and closed sections, necessitating the emergency dewatering of active portions in 1993. Based on bedrock strike in Leading Creek, regional groundwater (plan-view) flow direction is expected to trend from northwest to southeast. Hillside drainage aspects were not incorporated in assigning aereal extent of surface or underground mines to a particular drainage, where only rainfall fall-line analysis was used.

Except possibly for small subsheds like TS16, incorporation of bedrock strike would not be expected to greatly affect results reported here.

Summary History of Coal Mining in Leading Creek

Excluding historic and current SOCCO operations, most underground mining in the Leading Creek vicinity was conducted prior to 1910, with some abandoned mines created in the 1930's and after WWII. Post WWI and WWII underground mining was tied in some cases to strip mining activities that became increasingly active from the 1930's onward. Based on records at ODNR, coal was reportedly shipped from Pomeroy, Ohio as early as 1833. Though minimal in comparison to later activities, there was likely some localized underground mining in Leading Creek prior to the Civil War. Most underground mining in Leading Creek though occurred in the latter portions of the 19th century.

Most strip mining occurred after WWII based on interviews with watershed residents that indicated a significant increase in these activities in the 1950's. Based on aerial photo revision dates, 80% or 1379 ha of strip mines in Leading Creek were created by 1961, and 20% or 353 ha between 1961 and 1978. Little surface mining occurred after this period due to the onset and burdens of increased financial and environmental protection imposed by the Surface Mining and Reclamation Act of 1977. Excluding Thomas Fork where ASML was not digitized, only 4.2% by area overlap of ASML and AUML coverages was found in subsheds with AUML>0%. AUML was centered below ridges and ASML followed outer edge contours.

The typical case was that strip mining occurred after underground mining unless an abandoned highwall created by strip mining was further excavated using underground mining techniques. Near-surface underground and strip mining in Leading Creek were dominated by the excavation of Redstone #8A coal referred to as the Pomeroy #8A in Ohio, named after the Town of Pomeroy on the western shore of the Ohio River. The Redstone #8A formation is part of the Monongahela Bedrock Group, an estimated 90m thick sequence of non-marine sandstones, siltstones, shales, and coal with estimated age of 295 million years. The Monongahela Group would be expected to offer overall low to medium carbonate buffering capacity (USEPA, 1980).

Minor amounts of coal were likely extracted from the lower Conemaugh Group. Below this formation, the SOCCO deep mines work the Clarion #4A coal seam. This seam is part of the Allegheny Formation, a 100m thick assembly of marine sandstone, siltstone, shale, limestone, and multiple coal units with estimated age of 330 million years. Due to its marine origins, higher sulfur and alkalinity might be expected in the Allegheny Formation compared to surface lithologies (Caruccio et al, 1974; Younger, 1996), but on average this did not hold true for coal deposits in the nearby vicinity of Leading Creek (USGS, 1998). In Figure 3.7, SOCCO mine operations are still active and all underground mine extents are represented "as-mined", incorporating the outline of excavation and points of access in permit data.

Summary of Background Watershed Ecological Risk

Unglaciated itself, Leading Creek is a remnant of the original Teays River (Lucht *et al*, 1985). Glacial meltwater and outwash (Lucht *et al*, 1985; Feusner *et al*, 1989) produced Leading

Creek and forms the existing gently sloping valley network seen today. In the north portions of the watershed near Albany, Ohio extensive preglacial terraces sit atop gently sloping high level land. Leading Creek's northern and western headwater drainages, above and including Dexter Run's headwaters, are also direct remnants of the Teays River drainage system where preglacial terraces also characterize these areas. Representing the strip mining district, lower tributaries of Leading Creek are typified by long and steep hills, steepest adjacent to the Ohio River (Feusner *et al*, 1989). Leading Creek's average fall is 0.16% (100*km/km) (OEPA, 1991b). Mainstem slopes of the stream bed range 1% to 0.01%, from headwaters at Albany, Ohio to the mouth of the Ohio River, respectively. In summary, Leading Creek drains moderate to steeply sloping uplands and gently sloping lowlands (Gillmore *et al*, 1991). Due to thin valleys, the floodplain landscape is most typified by agricultural activities concentrated relatively close to watercourses.

Examination of aquatic habitat, toxicity, and biodiversity was conducted as part of a multi-disciplinary hydrologic-based ecological risk assessment of Leading Creek, located in southeast Ohio. Extensively sampled by others since 1993, researchers at Virginia Tech monitored this watershed between 1995 and 1998. The study was precipitated by a regulated emergency 28 day dewatering event of an active coal mine, the Meigs Mine No. 31, owned by Southern Ohio Coal Company (SOCCO) and parent company American Electric Power (AEP). In the summer of 1993, approximately 50 MGD of AMD with a pH 2.5 to 4.5 was discharged into Parker Run, a tributary of Leading Creek (Birge, 1995; Maloney, 1993; OEPA, 1994). The AMD had been stored in an inactive portion of Mine No. 31 awaiting treatment. A shaft seal failed resulting in flash flooding of the adjacent active portions of the mine whose continued operation was deemed critical to the local economy. By early 1996 natural recovery from the dewatering event appeared to be well underway, and relatively complete in Leading Creek with respect to historic and current land uses unrelated to the one time dewatering event.

Ecological risk assessment of mining and other land uses was carried out in various field surveys to identify, design, and implement a set of prioritized stream improvement projects throughout the entire watershed. This was to be quantified so that actions selected and taken would most improve mainstem biodiversity scores used to assess aquatic insect and fish community health (Karr, 1991; OEPA 1989; USEPA, 1980, 1989, 1992, 1995). Separate from restoration activities due to dewatering, the basis for this action was to provide guidance to federal regulators on how best to administer a $2 million stream improvement fund.

A multidisciplinary approach was chosen to directly assess stream health at each of 30 monitoring sites. This incorporated all known and suspected point and nonpoint source releases along the 54.6 km mainstem that drains 38,800 ha prior to its confluence with the Ohio River, near Middleport, Ohio. Background investigations indicated some reaches with decimated biodiversity. Areas in worst decline appeared most impacted by abandoned mining land uses and AMD. Minimum pH's ranged down to 3.3 in smaller 1st and 2 order subsheds and upwards of 3.7 to 5.0 in the largest, most heavily AMD impacted 4th order tributary. The pH values found in Leading Creek represent common levels for AMD mixed in similar size streams (Caruccio et al, 1974; Evangelou, 1995; Hedin et al, 1994; Nordstrom, 1982; USEPA, 1980). Semi-quantitative conclusions derived from the initial risk assessment were that AMD depressed biodiversity on several lower mainstem tributaries and at the mouth of Leading Creek below Thomas Fork. Sedimentation from both agricultural and AML were also determined to be deleterious in most tributaries and along the entire mainstem. Sedimentation was deemed more amenable to potential overall stream improvement given available funding (Cherry et al, 1999).

AUML Mining Technique and the Occurrence of AMD Instream

Because the Redstone #8A coal bed is associated with both surface and near-surface underground mining in Leading Creek, mining technique appears to play a functional role in the expression of AMD. An explanation of results for ASML and AUML in Leading Creek however could not be found in chemical characteristics of the Monongahela Bedrock Group. The Leading Creek Redstone #8A coal bed in the ASML mining belt near Rutland, Ohio, lies below a shallow layer of shale, which lies below a massive sandstone cap rock approximately 20 m thick. Going westward away from the Ohio River, the upper sandstone reduces to 6 m in thickness on the eastern side of the Salem Township. This characterization is based on unpublished measured sections down to coal bench elevations, recorded by ODNR, and was supported by local soil survey descriptions of bedrock and regional classifications (Gillmore *et al*, 1991; USEPA, 1980).

Published surface lithology is well described in Belmont County, 150 km northeast of Meigs County, along the Ohio River. This line tracks the mining belt of eastern Ohio where the Redstone coal bed runs east through West Virginia and northeast to Pennsylvania. Underscoring the variable nature of coal bed environments, in Belmont County, Ohio the Redstone bed is sandwiched between two substantial limestone deposits. The upper Fishpot Limestone occurs as a 6.4 -9.4 m thick sequence, and the lower Redstone Limestone is approximately 2.5-3m thick and can range in thickness up to 5.5m (USGS, 1963). Limestone in Leading Creek however is not expressed in the non-marine Monongahela Bedrock Group. Outside of the Leading Creek ASML belt, the lower Conemaugh Bedrock Group exhibits only minor amounts of limestone moving northwest. The presence of near-surface limestone in Leading Creek becomes substantial only at the upper northwest tip of the watershed. This occurs where the Brush Creek Limestone was historically mined, as shown in Figure 3.1 for the closed quarry. This relatively thin limestone layer is still mined today just outside the watershed boundary, west of the Town of Albany.

An important feature of coal bed lithologies in this region is the common occurrence of a clay/shale layer on the bench floor of mined coal seams. In the case of AUML in Symmes Creek near the Oak Hill, Ohio kilns, clay deposits were sufficiently thick to make co-extraction profitable. In Leading Creek underground mining permits indicated clay was not extracted with coal. Tying in structural lithology, pyretic sulfide bearing rocks that cause AMD are usually found in close proximity to the coal bed, often occurring in and around the coal seam itself (Skousen, 1996). In Ohio, coal deposits are characterized as “dirty” for this reason, creating the need to wash coal prior to its delivery to market as SOCCO does in Leading Creek. Compared to increased permeability of the disturbed seam, vertical transport of AUML derived AMD is likely regulated by the occurrence of clay aquitards below the bench floor. Geology in Leading Creek is described by horizontally oriented bedrock striking northeast-southwest with average dip of 5.7 m/km towards the southeast (Lucht *et al*, 1985). AMD transport to stream drainages would preferentially flow along dip of the disturbed bedrock, resulting in acidic discharge seeps at the southeastern most bench elevation. These seeps may re-enter the valley soil column after being discharged at hillside locations.

Structural Aspects of ASML and AUML in Leading Creek

In contrast, strip mining in southeastern Ohio typically removes coal and overburden down to a clay/shale bench floor, leaving the bench exposed. The same pyrite bearing materials found in AUML discharge AMD just above hillside-valley toes where AMD from AUML can manifest as seepage. For unreclaimed strip mines, post-blasting pyretic materials typically reside at or below grade of the original bench floor and were more widely distributed at the hillside toes. ASML results in a landscape characterized by large footprints with highly permeable and extremely acidic soils (Gillmore *et al*, 1991; Lucht *et al*, 1985). AUML surface waste piles, again as a matter of economy, remain less distributed than ASML, lying in close proximity to mine entrances. A lack of significance in correlations between Mine Gob and Mine Spoils and %AUML was observed for all sites. This was also true considering only sites with AUML>0%. Despite this geodata insensitivity, most or all AMD in subsheds with AUML>2% though may not necessarily derive from underground oxidation in exposed seams. During underground mining, substantial amounts of pyrite in Leading Creek were removed to and remain disposed at the surface as gob and spoil.

The resulting disturbed bedrock structures for AUML and ASML imply several things. For both ASML and AUML in Leading Creek, floor and ceiling clastic materials per unit area disturbed would be expected to have similar net maximum potential acidity and particle size characteristics (MPA) (Daniels *et al*, 1995; Skousen, 1996). This assumes a perspective from the streambed and homogeneous, chemically similar undisturbed coal beds. Buffering capacity potential for surrounding rock systems should also be similar where lack of carbonate materials characterizes both waste pile systems and downgradient paths. Homogenization of floor and ceiling materials with ASML overburdens in Leading Creek may be significant but does not offer direct explanations for reduced acidity per unit area of mining ASML observed at the stream bed. Physical reclamation of ASML was also not a significant factor; thus final cover and degree of homogenization of ASML spoils may also be insignificant.

Disposition of ASML and AUML Residues in Leading Creek

Final coal seam bench status (open = ASML, subterranean = AUML) and concentration of waste piles are two variables still differentiated in unreclaimed ASML and AUML pyretic waste systems. The hydrology of subterranean coal beds is substantial due to acidic seepage observed. Deeper zones in concentrated gob and spoil piles have also been implicated as discharge points for severe AMD (Daniels *et al*, 1995), but to the degree these outweigh acidity production in Leading Creek is unknown. Lack of correlation with %AUML and %Mine Gob and %Spoil were observed where %AUML was highly correlated to low pH instream. A relatively less studied aspect remains as to the chemical and hydrologic disposition of shallow, more widely distributed ASML waste piles that presumably store similar amounts of pyrite. These systems would appear to encompass different phenomena with respect to rates of reaction and acidic, reducing conditions necessary for the generation of AMD (Nordstrom, 1982; Skousen, 1996; Williamson *et al*, 1994).

ASML disturbance in Leading Creek is roughly ½ the age of major AUML disturbances. A question not answered here is the level of oxidation that has taken place to date for each solid waste system. Perhaps ASML materials have fully oxidized and leached, and perhaps comparatively more ASML derived AMD is transported in runoff during high flows. In work by Meek (1996) looking at several prevention technologies associated with surface mining techniques in Appalachia, it was indicated that mineralization and acid generation rates vary with time, peaking in surface generated waste piles after 4 to 8 years. The rate of pyrite oxidation and acid production is thought to be highest in the oxygenated surface layer of waste piles (Daniels *et al*, 1995). Daniels indicated that due to total pyrite mass and relatively slow rates of water movement through coal mining waste piles, time frames for significant acid generation would be expected to last for decades if not longer.

Literature Cited

See Chapter 2 for Literature Cited

Appendix B - Biodiversity and Habitat Data

Justin Eric Babendreier

Table B1 – Leading Creek Benthic Macroinvertebrate Averaged Results; 1995 to 1997 Spring Through Fall Surveys. Standard deviations are given in parenthesis. Tributary and mainstem sites sampled twice in 1996 (EPT 2x); Parker Run and all mainstem sites three sampled times in 1997 (EPT 2x); and sites below Mine 31 sampled twice in 1995 (no EPT data). Standard deviations are given in parenthesis. Community strength defined given by $S_{Comm.}$ $=Taxa^{a}$ + Abundance; with a = 2 for EPT data and a = 1.5 for all insects. . LSC5A was used for additional biological survey collections and is located upstream from station LCS5B.

Station ID	Total Average		EPT Average		Community Strength	
	Abundance	Taxa	Abundance	Taxa	Total	EPT
(R)FCS1	208 (2.1)	30.5 (3.5)	87.5 (50.2)	9.5 (0.7)	376 (27)	178 (64)
(R)SCS2	- -	33 -	- -	10 -	- -	- -
LCS1	133 (43.9)	25.7 (7.2)	82.0 (38.2)	10.0 (1.4)	265 (86)	183 (66)
LCS2	139 (36.4)	26.7 (6.5)	76.5 (34.6)	12.0 (1.4)	279 (26)	222 (0.7)
LCS3	76.7 (9.2)	23.7 (5.5)	15.0 (8.5)	6.5 (3.5)	193 (41)	63.5 (54)
LCS4	124 (26.6)	25.3 (2.5)	61.5 (24.7)	8.5 (2.1)	252 (16)	136 (61)
LCS5A	131 (64.1)	32.0 (7.8)	54.5 (17.2)	9.5 (1.0)	316 (122)	146 (37)
LCS5B	45.7 (36.3)	15.7 (9.6)	17.0 (17.0)	4.3 (2.8)	116 (88)	40.8 (39)
LCS6	84.7 (55.6)	21.1 (11.1)	27.0 (23.4)	4.5 (2.9)	190 (132)	53.5 (49)
LCS7	267 (167)	35.6 (11.2)	52.8 (28.2)	6.8 (2.9)	486 (250)	105 (65)
LCS8	61.0 (55.1)	19.1 (13.3)	12.5 (9.8)	4.5 (3.1)	158 (138)	40.0 (27)
LCS9	74.6 (42.7)	22.3 (10.5)	14.0 (10.5)	5.0 (2.9)	189 (102)	45.5 (40)
LCS10	31.1 (21.4)	12.9 (7.5)	11.0 (14.2)	4.0 (2.6)	83 (58)	32.0 (35)
TS1	94.0 (25.5)	23.5 (0.7)	2.0 (2.8)	2.0 (2.8)	208 (20)	10.0 (14)
TS2	96.5 (50.2)	27.0 (7.1)	2.0 (1.4)	2.0 (1.4)	239 (105)	7.0 (7.1)
TS3B	112 (59.4)	24.0 (8.5)	9.0 (2.8)	3.5 (0.7)	232 (122)	21.5 (7.8)
TS4	128 (66.5)	30.0 (2.8)	15.0 (7.1)	6.5 (2.1)	293 (43)	59.5 (35)
TS5	88.0 (86.3)	25.0 (11.3)	23.5 (24.7)	4.0 (4.2)	218 (171)	48.5 (59)
TS6	159 (4.9)	28.5 (0.7)	47.0 (28.3)	6.0 (2.8)	311 (11)	87.0 (62)
TS8A	69.0 (25.5)	16.0 (8.5)	23.5 (2.1)	3.5 (0.7)	136 (76)	36.0 (2.8)
TS8B	53.5 (6.4)	20.5 (6.4)	5.5 (3.5)	3.0 (1.4)	148 (37)	15.5 (12)
TS9	60.5 (0.7)	23.5 (3.5)	17.5 (12.0)	5.0 (1.4)	175 (26)	43.5 (26)
TS10	56.0 (43.8)	17.5 (4.9)	28.5 (34.6)	4.5 (2.1)	130 (75)	51.0 (54)
TS11	137 (89.0)	28.0 (10.3)	22.0 (16.1)	3.8 (1.7)	316 (130)	40.1 (30)
TS12	72.0 (17.0)	21.0 (5.7)	28.5 (17.7)	4.5 (2.1)	170 (56)	51.0 (37)
TS13	72.5 (21.9)	24.0 (7.1)	15.5 (16.3)	3.5 (0.7)	192 (74)	28.0 (11)
TS14	73.5 (3.5)	18.0 (4.2)	23.5 (4.9)	5.5 (3.5)	151 (23)	60.0 (34)
TS15	24.0 (18.4)	9.0 (1.4)	3.0 (1.4)	2.0 0.0	51.1 (25)	7.0 (1.4)
TS16	133 (115)	10.0 (4.2)	2.0 (2.8)	0.5 (0.7)	165 (95)	2.5 (3.5)
TS17	7.5 (9.2)	3.5 (3.5)	2.5 (3.5)	1.0 (1.4)	15.3 (19)	4.5 (6.4)

Table B2 – Leading Creek Qualitative EPA RBP Habitat Survey Results - 1996. Reach-level data based on USEPA's Rapid Habitat Bioassessment Protocol (USEPA, 1989).

Station ID	ASML	AUML	No.2	No.31	AG	Urban	Substrate /Cover	Embeddedness	Flow/Velocity	Channel Alteration	Scour /Deposition	Pool/Riffle-Run/Bend	Bank Stability	Bank Vegetation	Streamside Cover	Total
(R)FCS1	x	x			x		14	10	16	10	10	11	5	9	7	92
LCS1					x	x	6	14	10	14	14	8	3	5	10	84
LCS2					x		16	8	12	14	9	9	9	10	10	97
LCS3					x		0	0	2	0	0	2	5	6	3	18
LCS4			x		x		8	4	8	2	1	5	1	4	4	37
LCS5B	x		x		x		15	18	10	7	9	5	1	1	8	74
LCS6	x		x	x	x		15	12	13	13	10	13	3	4	7	90
LCS7	x		x	x	x		16	7	18	7	5	14	6	7	7	87
LCS8	x	x	x	x	x		6	3	11	1	1	5	4	4	7	42
LCS9	x	x	x	x	x		0	0	13	1	3	10	2	1	1	31
LCS10	x	x	x	x	x		5	5	3	1	2	2	7	9	7	41
TS1					x	x	1	1	2	1	2	1	9	10	9	36
TS2					x	x	3	4	3	1	2	3	9	10	10	45
TS3B					x		2	1	3	2	2	2	5	4	10	31
TS4					x		1	1	6	1	0	5	9	10	5	38
TS5					x		1	0	1	0	1	0	1	4	1	9
TS6			x		x	x	6	2	14	2	1	11	1	2	6	45
TS8A	x				x		1	1	5	1	1	2	8	8	5	32
TS8B	x				x		9	1	4	2	3	3	5	7	7	41
TS9					x		3	2	3	2	2	2	8	9	10	41
TS10	x				x		4	2	3	4	4	4	8	9	8	46
TS11				x		x	13	15	16	11	8	11	10	9	7	100
TS12					x		3	4	7	5	3	8	2	3	4	39
TS13	x				x		6	4	4	2	4	5	8	9	9	51
TS14	x	x			x	x	1	0	2	1	1	2	7	9	5	28
TS15	x	x			x		8	4	10	2	2	8	4	8	9	55
TS16	x	x			x		7	3	11	5	5	7	8	10	10	66
TS17	x	x			x		5	3	8	1	1	6	2	3	3	32

Table B3 – Leading Creek Qualitative QHEI Habitat Survey Results - 1996. Site-level data based on Ohio EPA's Qualitative Habitat Evaluation Index (QHEI) (Rankin, 1989).

Station ID	ASML	AUML	No.2	No.31	AG	Urban	Substrate	Cover	Channel	Riparian	Pool	Riffle	Gradient	Total
(R)FCS1	x	x			x		14	11	20	7.5	9	6.5	10	78
LCS1					x	x	12	6	14	7.5	7	6	10	62.5
LCS2					x		13	13	18	8	5	6	8	71
LCS3					x		8	6	10	4	9	0	8	45
LCS4			x		x		11	12	14	7	8	4	8	64
LCS5B	x		x		x		16	7	16	4.5	7	3	6	59.5
LCS6	x		x	x	x		12	9	17	5	9	2	6	60
LCS7	x		x	x	x		14	12	19	5	8	5	6	69
LCS8	x	x	x	x	x		6	4	10	4.3	8	2	8	42.3
LCS9	x	x	x	x	x		6	5	12	4	3	0	8	38
LCS10	x	x	x	x	x		9	3	11	6	0	0	8	37
TS1					x	x	0	10	11	6	5	0	8	40
TS2					x	x	2	10	12	6	5	0	10	45
TS3B					x		9	8	12	6.3	4	0	10	49.3
TS4					x		9	6	15	6	6	0	8	50
TS5					x		9	2	11	4.5	0	0	8	34.5
TS6			x		x	x	10.5	6	13	6	7	4	8	54.5
TS8A	x				x		9	7	11	4.5	0	0	8	39.5
TS8B	x				x		8	8	12	7.3	0	0	8	43.3
TS9					x		9	11	12	8.8	8	0	8	56.8
TS10	x				x		8	13	14.5	3	7	0	4	49.5
TS11				x		x	12	8	16	7	7	3	4	57
TS12					x		8	6	13	4.5	7	1	6	45.5
TS13	x				x		11	5	14	4.5	0	3	6	43.5
TS14	x	x			x	x	6	5	10	6.5	0	0	8	35.5
TS15	x	x			x		7	11	14	7.1	4	0	6	49.1
TS16	x	x			x		7	10	15	6	3	0	6	47
TS17	x	x			x		7	9	12	3.3	4	2	6	43.3

Appendix C - A Hydrologic-Hydraulic Based Flow Regime Modeling Framework for Watershed-Scale Ecological Risk Assessment

Justin Eric Babendreier

Appendix C – A Hydrologic-Hydraulic Based Flow Regime Modeling Framework for Watershed-Scale Ecological Risk Assessment

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(ABSTRACT)

A methodology is presented that allows quantification of hydraulic watershed characteristics on temporal and spatial scales for use in ecological risk assessments and water quality monitoring programs.

- A geomorphologic-based statistical methodology was developed to successfully model drought, low, medium, high, bankfull, and flood flow regimes using multiple gages distributed across small to large subsheds.
- On Leading Creek, the methodology enabled quantified separation of flow-variant ecological risk assessment data sets into hydrologically meaningful (low, medium and high) instream flow conditions.
- Flow regime data separation was accomplished using a centralized single stream gage and a multiple-gage approach.
- For instream propeller-type velocity meters used in stage-discharge rating curve development, velocity meter response was shown to be strongly non-linear with respect to velocity, invalidating standard calibration approaches.
- Above 0.2 m^3/s velocity meter response tends to become invariant to velocity. Below 0.2 m^3/s a steeply sloped linear function of velocity can be used to adequately describe a correctable dependency between instream velocity and actual meter response.

Appendix C - A Hydrologic-Hydraulic Based Flow Regime Modeling Framework for Watershed-Scale Ecological Risk Assessment

INTRODUCTION

Analyzing risk assessment data sets in light of base flow and runoff conditions can provide critical insight into the hydrologically driven relationships that exist between human activity on the surface, it's expression instream, and its impact upon aquatic life. The importance of a well-defined hydrologic-hydraulic framework was investigated here for application to watershed-scale studies of common anthropogenic stresses in Ohio, including underground mining and abandoned surface contour mining or strip mines (ASML). Together with urbanization in small subsheds and poor agricultural practices, severe sedimentation is a ubiquitous problem plaguing many of Ohio's landscapes. Coal mining though represents the majority of impacts seen today in the Leading Creek Watershed located in southeastern Ohio. Abandoned mined lands (AML) have been shown to cause significant degradation to both habitat and water quality. Originating from abandoned underground mine land (AUML), uncontrolled, untreated pyretic acid mine drainage (AMD) reaches Leading Creek today. Originally disturbed almost 100 years ago, the dominating legacy of near-surface AUML appears to be most responsible for decimation of its aquatic communities. To further study this stream, a thorough review of its hydrology and hydraulics was completed.

An expansion of watershed data sets and hydrologic models is offered, building upon previous assessment in the Leading Creek Improvement Plan (Cherry et al, 1999). Developed for American Electric Power and the Southern Ohio Coal Company (SOCCO), the improvement plan was associated with an emergency dewatering event in 1993 that released over a billion gallons of AMD from a closed portion of a deep active underground mining operation. This paper focuses on development of a quantitative hydrologic-hydraulic analysis of Leading Creek to support further quantitative-based risk assessment of the stream system. By accounting for flow regime, land use data can begin to be brought into discernable focus by allowing for a more detailed description and accurate analysis of instream pollutant concentrations. Chemistry data segregated by hydrologic-hydraulic condition leads to significant understanding of mechanisms and processes. In lieu of direct estimation or measurement of instantaneous flowrates for water quality sampling programs, flow regime itself can provide key insights, reducing lingering equivocation between land use and the temporal distribution of instream pollution. This paper describes the technical basis underlying collection and use of hydrologic-hydraulic data, development of flow regime models, and their application to water quality monitoring results for ecological risk assessment of Leading Creek.

C.1.1 Flow Regime Modeling For Ecological Risk Assessment

In flow modeling we are interested many times in assessing relatively steady state loading patterns from point and nonpoint sources distributed across a variable hydrology in space and time. For AMD pollutant loads arriving instream via both base flow and runoff, a constant load assumption will always need to be qualified in some circumstances because of significant inter-annual cycle offsets that can be experienced between groundwater and surface water. Nonetheless, AML and associated acid mine drainage represent an ideal study environment for application of statistical flow modeling concepts for ecological risk assessment. Water flow regime has been distinguished as an important element of biotic integrity (Karr et al, 1986; Yoder et al, 1996; USEPA, 1996c). Described in the context of integrated ecological risk assessment, Karr and later Yoder and Rankin highlighted five major environmental effects that distinguish protection and classification of biotic integrity and water resources: a) energy source, b) water quality, c) habitat structure and quality, d) flow regime, and e) biotic interactions. Flow regime defined by Karr (1986, 1991) underscored effects from water volume, temporal distribution of floods and low flows, and flow regulation. Yoder and Rankin (1996) significantly expanded concepts of flow regime through re-classification, establishing significant effects from groundwater, precipitation and run-off, velocity, land use, and high-low extremes.

These formulations are captured to degree by early work of Milhous (1998) and Wesche who pioneered flow regime as a critical function of habitat assessment through hydraulic modeling for fisheries sciences. Developed to assess ecology, the fisheries based Physical Habitat Simulation Model, PHABSIM, directly relates hydraulic features of a reach to biotic production. The model PHABSIM defines a method for evaluating the availability of physical microhabitat in various streams under different hydraulic scenarios (Wesche *et al*, 1980). It is a physical model, and does not directly evaluate chemical impacts. The computer model is based on the Instream Flow Incremental Method (IFIM), and can incorporate flow depths, velocity, substrate and cover preferences of fish. The model is one of few available that relate hydraulic aspects directly to biodiversity in a simulation setting. The growing importance of connecting hydraulics to ecology is further evident in the rise in contemporary microhabitat measurement, where in-depth velocity profiles are being used to assess fish habitat preferences (Beebe, 1996).

Demand for integration of user-friendly basin models such as HSPF may eventually lean towards ecology. This will occur in response to a need to facilitate indepth simulation of ecological and ecotoxicological conditions (Cairns et al, 1991, 1996; Donigian et al, 1995). An interesting question remains as to how can we best bring together classical hydrologic simulation and classical ecological risk assessment. Increases in demand for “integrative” type assessments continue to follow the rising demand and value we place on clean water, controlled erosion, and flourishing biodiversity. Perhaps future investigations will arrive at some combination of applications like PHABSIM and HSPF via a platform like BASINS that can integrate Geographical Information Systems (GIS) (Aronoff, 1993; Donigian, 1995; Wesche et al, 1980).

Watershed management is more necessary than ever in limiting man’s activity on the landscape where this activity may otherwise jeopardize water intake plants, navigation, or recreation potential (Bonta et al, 1992; Broner et al, 1996; Curtis, 1976; Kershner, 1997; Lenat, 1984, 1993; Thomann et al, 1987; USDA, 1983, 1985). Increasing focus on water and biological resources management will give additional recognition to aquatic ecology and the rising value society continues to assign to watersheds (Cairns et al, 1996; Johnson, 1995; Minshall et al, 1985; NRC, 1992; Suter, 1998; Vannote et al, 1980; Waters et al, 1995; USDA, 1998). A convergence towards integral, quantitative study of hydrology, hydraulics, and ecology, termed here as hydroecology, seems inevitable in watershed management sciences and engineering.

C.1.2 Flow Regime Modeling Study Objectives

Flow regime is interchangeable with flow condition. The purpose of this work was to:

1) Define, create, and adequately document a sound technical basis for flow modeling and hydrologic-hydraulic characterization of risk assessment data sets in Leading Creek,

2) Develop, evaluate, and demonstrate the applicability of statistically based static-condition flow models for Leading Creek across low and high flow extremes that can be used to quantify flow regime in a manner consistent with geomorphologic principles,

3) Accomplish flow-based data separation tasks within a context meaningful to past and present land uses, and that facilitates direct insight into familiar engineering concepts and principles related to the transport and fate of pollution moving through the landscape.

The utility of a watershed-scale water flow modeling methodology was investigated for application to ecological risk assessments. The methodology presented here addressed aspects of hydrologic-based risk assessment by employing GIS based land use/cover analysis in conjunction with standard techniques for velocity and flow measurement. The objective was to provide a well-defined hydrologic-hydraulic water flow modeling framework for evaluating quantitative water quality data associated with agriculture, urbanization, surface mining, and active and abandoned underground coal mining land uses. The purpose of flow modeling in Leading Creek was to facilitate an improved understanding of impacts to instream sediment and water chemistry, toxicity, and biodiversity. This was accomplished on two levels. On the simplest level it was desirable to retain a capacity to segregate flow-variant water quality monitoring data into groupings reflective of hydraulic dilution concepts. Concepts of low, medium and high flow were initially devised to distinguish between hydrologic processes associated with baseflow (or groundwater) discharge and overland run-off. On a second more detailed level, ability was also sought to accurately predict instantaneous flow rates across the watershed for water quality monitoring events scheduled over a two-year period. The aspect of instantaneous flowrate modeling for water quality monitoring is addressed in Appendix D.

Flow condition modeling was intended to provide a sufficient level of discrimination in assessing hydraulically driven mechanisms among land uses that may contribute to declines in biodiversity. Detailed watershed-scale hydraulic characterization for ecological risk assessments are not common in literature. This is particularly true for studies that employ Geographical Information System (GIS) at fine scales of 1:24000, land use/cover spatial analysis, and ecotoxicological based risk assessment. Hydrologic-hydraulic modeling reported here was developed as supporting tasks for an entailed multi-disciplinary risk assessment of a large watershed (388 km^2) employing those facets. This paper did not test a specific hypothesis other than demonstrating the applicability of well-understood hydraulic engineering principles to ecological based risk assessment. In efforts eventually undertaken, a framework of drought, low flow, medium flow, high flow, bankfull flow and flood regimes were proposed.

A technical note also investigated here was related to use of propeller type velocity meters in flowrate characterization. This was based on an observation that typical propeller-type velocity meters exhibit highly non-linear responses with respect to magnitude of velocity.

C.2 STUDY WATERSHEDS

Leading Creek is a 5^{th} order stream (scale: 1:24000) spanning primarily Meigs County, and portions of Athens, and Gallia counties in Ohio. Together, 11 mainstem stations and 19 tributary stations were selected for routine hydrologic, hydraulic, and ecological risk assessment monitoring. The monitoring network is detailed in Figure 3.1 in Chapter 3. Mainstem stations were located to provide complete longitudinal coverage of Leading Creek, and discrimination among potential sources of sediment or contaminant impacts to the stream system.

C.2.1 Leading Creek Mainstem and Tributary Stations

Ten mainstem sites and six of the tributary sites were included in stream stage monitoring indicated by designation as staff gage stations in Figure 3.1. Two continuous gage stations at mainstem sites LCS5B and LCS9 were continuously monitored for various parameters on a 5-minute time step. CGS data collection efforts provided continuous tracking of several water quality indicators describing instream environmental conditions throughout the study period. LCS5B is just above Parker Run's confluence and LCS9 is above Thomas Fork's, as shown in Figure 3.1. The two continuous gage stations LCS5B and LCS9 reflect significant differences in character between upstream and downstream mainstem landscapes. Agricultural activities dominate the upper mainstem while downstream reaches are dominated by mining.

Eighteen tributaries in the Leading Creek Watershed received some level of consistent sampling effort for most variables described here. These stations were initially chosen based on association with either (1) their known or suspected role in the contribution of point source pollutant loading to Leading Creek, or (2) their significance based on the size and nature of associated drainage area. The six tributaries selected for stream stage monitoring were chosen based on proximity to significant points on the mainstem of Leading Creek, proximity to known point sources and suspected nonpoint sources, or based on their significant representation of major tributary inflows. Parker Run (TS11) and Ogden Run (TS6) in Figure 3.1 receive active coal mining treatment plant effluents and assorted stormwater and sanitary discharges. Other NPDES sites relate to sanitary treatment (STP), water treatment (WTP), and quarry discharges.

C.3 METHODS AND MATERIALS

Data collection in Leading Creek and the reference watersheds included monitoring of: (1) Land use and land cover, (2) precipitation, stream stage, water velocity, and flow; (3) sediment and water quality; (4) benthic macroinvertebrate and stream habitat; and (5) toxicity through sediment, acute, and *in situ* testing programs. To assess chemical, physical, and biological interactions, stations in Figure 3.1 were examined over a wide range of drainage areas and stream orders (Minshall *et al*, 1985; Strahler, 1957; Vannote *et al*, 1980; USEPA, 1996c). Data collection methods specific to measurement of absolute vertical elevations of stream surfaces, slope, stage, water velocity, rainfall, flow and drainage basin analysis are detailed here including methods used for water quality indicators.

C.3.1 Land Use and Land Cover

Spatial analysis of subsheds used several sources of spatial data including data hand-digitized from USGS quadrangle 7.5-minute map series (scale 1:24000), field data, and USGS hydrology geodata obtained from Ohio Division of Natural Resources (ODNR). Leading Creek subsheds were defined by digitizing hand drawn rain fall lines that also incorporated interpretation of the major influences of active drainage management associated with roadways. Initial conversion and processing of basemap ARC/INFO® exchange file formats for ODNR geodata was carried out using either ArcView® 3.0 and later versions or the Atlas® Import/Export program. Data describing locations of study sites, stream order, drainage area, stream length, and associated land use/cover characteristics were calculated using Atlas-GIS® Version 3.01 for Windows® and linked Microsoft Office® 97 spreadsheet-database programs. All statistical analysis was performed using the NCSS® 97 software package (Hintze, 1997) or routines in the Excel spreadsheet package. All raw and processed data was managed via linked ACSESS and DBASE III databases and pivot tables.

The range of drainage areas describing 61 primary tributaries and associated subsheds in Figures A1 and A2 of Appendix A span 37 to 8038 ha each, where the smallest subshed routinely monitored for water quality was 124 ha, and the smallest gaged subshed was 157 ha. See Table A1 in Appendix A for specific drainage areas for each routinely monitored subshed and quantified geographic locations of cross-sections studied. All GIS data and area calculations were standardized to geographic coordinates using the North American Datum (NAD) of 1927 and retain horizontal positional accuracy ± 15m.

C.3.2 Continuous and Manual Gage Stations

In Leading Creek, a technically sound basis was desired to provide hydrologic inference for water quality monitoring programs and aspects of risk assessment data analysis. System wide quantification of stream hydraulics was deemed to be an integral element during risk assessment design. Hydraulic monitoring was accomplished using two continuous stream gages that split the mainstem, and 14 other manual staff gages placed in small and large tributaries. Equipment at each continuous gage station in Figure 3.1 included an ISCO Model-4230 bubbler level meter and Model-675 automatically tipping rainfall gage. Continuous monitoring with automated data storage allowed comparison of stream stage (± 0.025 cm), flow (± 10 to 20%; estimated), and rainfall data (± 0.025 cm). Flow data accuracy was site dependent and regime dependent. A sampling frequency of 5 minutes was used for all parameters. Approximately monthly maintenance visits were conducted to calibrate instream equipment, to check rain gages, and to download continuous data collected onto a portable AT&T 386 lab-top computer.

Each continuous gage station was powered by two 6V Trojan 120 amp-hour golf-cart batteries wired in series, allowing 3 to 6 months before recharged-replacement units were switched-out, typically done on a monthly basis. The continuous meters were housed in 1.5x1.5x1.5m wooden houses located on second terraces and all interconnections buried in 5 cm PVC conduit, except rainfall gage wiring which was buried at shallow depth using heavy duty commercial wiring with UV protected coatings. Rainfall gages were set in open areas 3 m off the ground with cover from wind provided by tree lines > one height away. The instream transducers for bubble lines and water quality sondes were securely mounted instream, separate from each other. Conduit for sondes was exposed at the bank top, ran downstream along the bank, and was secured off the sediment bed with anchoring. This allowed access for retrieval and calibrations. Sand build up at station CGS2/LCS9 needed to be periodically cleared.

Southern Ohio Coal Company (SOCCO) personnel collected weekly stage readings throughout Leading Creek. SOCCO personnel also routinely monitored daily rainfall at their Mine No.2 Coal Office at Point Rock, Ohio and the Mine No. 31 site in Figure 3.1, referred to here as No. 2 rainfall and No. 31 rainfall, respectively. Process treated wastewater flowrates from Mine No.31 and Mine No. 2 shown in Figure 3.1 were also routinely monitored on weekdays for NPDES compliance monitoring and reporting. Manual staff gage control points (inverted guides) were constructed of 2.5 cm diameter PVC tubes, approximately 0.5 to 1.5 m in length, rigidly attached to a bridge guardrail. Static water level measurements were made with a Solinst-100 electronic water level meter with audio signal and environmental probe or a steel tape measure together with line-of-sight reading to the tape end and water surface, both with similar accuracy (± 0.03 cm). Each top of PVC measurement was eventually transformed to a professionally surveyed vertical elevation datum. Surveys tied each gage to a known benchmark above mean sea level with elevation accuracy of ± 0.03 to 0.3 m. Absolute vertical control for staff gages was achieved using dual-unit geographical positioning systems or rod and level. Prior to benchmark surveys, on-site vertical elevation control was established at start-up through two control points using an Pentax Pal - 5C autolevel and 13 ft wooden rod. As feasible, one spad each was placed on the bridge structure and one on a nearby mature tree or utility pole.

C.3.3 Channel Characteristics, Velocity, and Flow

Measured sediment properties, channel geometry, and observations in the streambed were used to characterize geomorphologic properties site to site. A combination of pebble counts, sieve analyses and hydrometer methods were selected for particle size analysis (Wolman, 1954; ASTM, 1990b, 1993; Gee et al, 1986). Limited sieve and hydrometer data were provided via routine sediment toxicity testing procedures establishing %sand, %silt, and %clay in samples. Stream stage height was calibrated to periodic velocity discharge measurements collected at each staff gage to allow for computation of stage-discharge rating curves (Bedient et al, 1992; Linsley et al, 1975; Yang, 1996). Stage-discharge ratings at each site were based on 5 to 16 data points collected approximately monthly at CGS sites and quarterly at other staff gage stations between April 1995 and June 1997. Each station rating included low flow data from system wide surveys conducted in September and October of 1996. One of the quarterly surveys was also conducted during an approximate 100-year flood, helping provide measurements across very low through high flow regimes. Added rating curve points were also considered and included a high-flow estimate derived from Leopold's bankfull equation $Q \propto (\text{drainage area})^b$. For gravel bed reaches, supporting hydraulic analysis was evaluated, incorporating pebble count particle diameter (D_{90}), reach geometry, and Manning's equation (Vanoni, 1977; Yang, 1996; Leopold et al, 1964).

Velocity measurements at 1 to 3 vertical depths were taken with a propeller type Swoffer Model-2100 velocity meter using some 5.1 cm propellers, but mostly 7.6 cm diameter propellers, each continually calibrated with still pond measurements. Matched rotor-propeller sets were always maintained where two rotors were used. Wading and bridge cross-sections were established at each site to allow profiling during low to high flow conditions. The propeller mounted on a graduated (3 cm) aluminum pole 7.3 m in length, collapsible in 0.91m sections, and approximately 3 cm in diameter. A Keson nylon surveyor's tape was used for width measurements (61 m; graduated 3.1 mm). Reference watershed stations were not included in stage-discharge rating activities. Flow computation followed the method of midsections (Bureau of Reclamation, 1984; Rantz, et al 1982). If encountered during higher flow, pole deflection angles of 5 to 30 degrees were estimated visually and corrected for later in data analysis. To maintain position instream, use of the bridgedeck and guardrails was often needed to brace the pole for currents > 0.6 to 0.75 m/s. All length measurements were recorded in English.

C.3.4 Water Quality Indicators

Water quality sampling was conducted at all mainstem, tributary, and reference sites shown in Figure 3.1. Monthly water column grab samples for pH, specific conductivity, and water temperature were collected and analyzed in the field, along with periodic monitoring for dissolved oxygen (DO). Water column samples used standard water and wastewater methods (APHA, 1992). Possibly not inclusive of all data collected, field meters included Corning Model-90 Checkmate multimeters, an Orion Model-122 conductivity meter, a YSI-55 oxygen meter, and a Fisher Accument Model-1003 pH meter. Specific conductivity was normalized to 25 ^{o}C (1.91% standard temperature coefficient; ± 1 umho/cm) and water temperature (± 0.01 ^{o}C). Portable pH probes were calibrated with pH 4, 7, and 10 lab-grade buffers and conductivity was calibrated with standardized KCl solutions using lab-grade deionized water (APHA, 1992). Stage at gage sites was routinely monitored during water quality sampling events. Indicator data was useful for understanding *in situ* dynamics from NPDES discharges shown in Figure 3.1.

C.4 RESULTS

C.4.1 Continuous Stream Flow and Water Quality Indicators

Figure D1 in Appendix D represents the duration of continuous gage monitoring activities at stations CGS1/LCS5B and CGS2/LCS9. Gaps in CGS records indicate backwater, malfunction or other data loss. March 1997 approximate 100-year flood peaks are not shown where CGS2 was disabled for this time period. Missing active mine daily data in Figure D1 indicates discharge rates < 0.01 m^3/s. Figures 5.2a and 5.2b in Chapter 5 give summarized data for pH and conductivity across Leading Creek. Specific conductivity highlights differences between AMD impacted reaches with moderate levels, Mine Nos. 2 and 31 and lower mainstem sites with highest levels attributed to treated AMD, and decreasing levels from LCS1 in the upper mainstem attributed to discharges from a closed limestone quarry operation.

In comparison to active underground mining, minor point source permitted National Pollutant Discharge Elimination (NPDES) discharges given by the WTP, STP in Figure 3.1 and sanitary wastewaters from the active mines represent insignificant loading of treated effluents and flow into the stream system. The active mining operations must meet categorical effluent limitations established for coal treatment (coal washing) plants. Coal preparation plant wastewater effluent is high in total dissolved solids (TDS) primarily composed of sulfate, sodium, and chloride where conductivity at SOCCO Mine No.31 typically ranges between 4000 to 6200 umhos/cm. During lower flow conditions the watershed can gain significant alkalinity from deep long wall mining mine operations, helping buffer AML-AMD entering the lower mainstem. The closed limestone quarry discharge is also a significant contributor of headwater alkalinity during low and medium flows. For most periods of flow, Mine No.2 wastewaters are conveyed to Mine No. 31 for treatment and storage before final discharge to Parker Run (TS11). Following periods of increased groundwater infiltration after run-off, some process wastewaters are treated and discharged to Ogden Run (TS6). To assist data interpretation, flow, rainfall, and water quality data at SOCCO Mines No. 31 and No. 2 from January 1996 through July 1998 were assimilated into the watershed database. Continuous gage stations went online 5/14/96.

C.4.2 Stream Elevation and Slope Models

Tables E1 through E4 in Appendix E presents gage station vertical survey control metadata. These were provided to document and record the details of hydraulic calculations. Stage and slope calculations were used here in several ways to arrive at final comparisons of flowrate and flow condition between stations. Vertical survey control served as the basis for defining the slope-energy profile of the stream system, including backwater conditions at the mouth. Stations LCS7, LCS8, LCS9, LCS10, and TS17 experience periodic backwater from the Ohio River controlled by hydroelectric dams and navigational lock structures. Table E1 gives initial survey data used to establish on-site survey control at each site. Table E2 provides summarized elevation survey data collected at each staff gage station (SGS) used to establish elevation referenced to mean sea level (AMSL). Table E3 provides an error analysis and comparison of the two independent surveys conducted at each SGS station. Table E4 provides a reconciled final benchmark survey control summary. Tables E2, E3, and E4 establish surrogate relative control points (i.e. alternate measuring points) at some sites needed to re-establish elevation control due to vandalized PVC pipes.

Weekly head measurements were collected through the entire period of watershed monitoring extending from 3/26/96 to 6/9/98, with added, independent events occasionally. From this data various hydrologic models were developed including stream elevation and slope models for various flow conditions in the system. This included minimum stage, average, and maximum stages encountered during routine monitoring and two major floods occurring in June 1996 and March 1997. Criteria for segregation of head data are discussed below. With the controlled survey data approach, the internal database system used bridgedeck readings of distance to the stream surface from a fixed point. In this way the relative measuring point ("MP") staff gage program could be integrated from site to site to establish energy slope "photographs" up and down the mainstem. This was done with a good deal of confidence in accuracy using 141 independent staff gage collection events, and other static water level data collected thorough water quality monitoring. Data management was reviewed for quality assurance of transcription, and screened for outliers.

Free surface elevation and differentiated slope models as a function of river kilometer were developed using best-fit polynomial equations and surveyed control elevation data. Shown in Figures C1, C2, and C3, from different curves developed analysis determined that slope was relatively stage-invariant. The curves followed expected concave shapes ($R^2 > 99.9$; n=11) (Leopold *et al*, 1964). Figures C1, C2, and C3 include Station TS8B that is 0.06 km from the mainstem confluence. Except for indicating the extent of backwater during the flood of 1997, figures exclude backwater station-events identified. Stream length data between stations was derived from the GIS

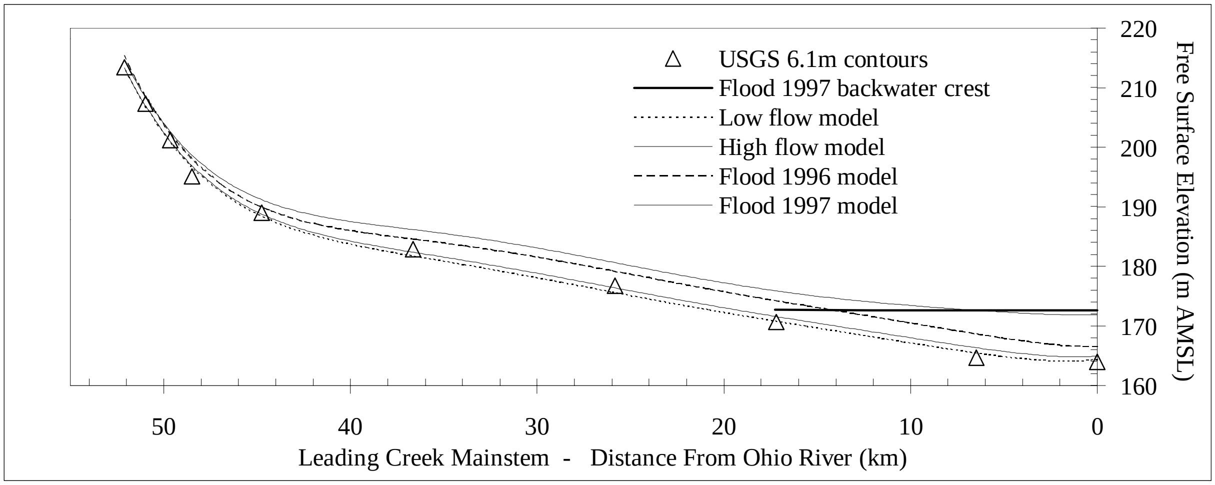

Figure C1 - Leading Creek Free Surface Elevation Data – Data represents statistics of weekly and monthly sampling, and USGS map data.

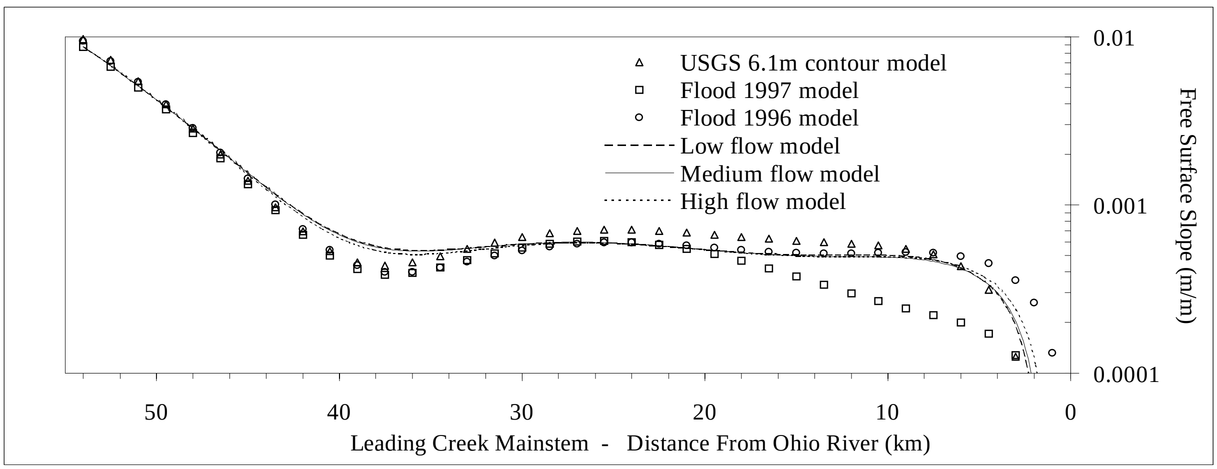

Figure C2 - Leading Creek Free Surface Elevation Flow Condition Models – 6th order polynomial curves fit to mean periodic sampling data.

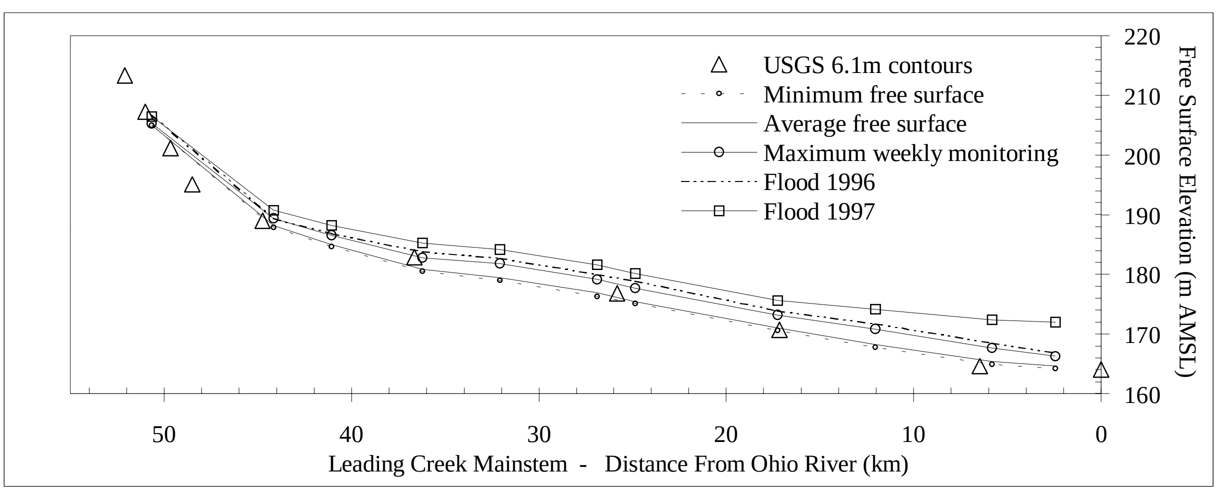

Figure C3 -Leading Creek Free Surface Slope Models – 5th order derivative polynomials curves.

along with locations of 6.1m contour crossings using USGS 7.5-minute quadrangle maps for comparison to surveys. Figure C3 includes a USGS 6.1m contour data model based on offsets using data from the nearest gage station for various stage-flow conditions considered. Stage elevation at tributary confluence points on the mainstem or any other interim point were easily derived from the models, allowing accurate determination of slopes on gaged tributary reaches for various flow conditions. Compared to minimum free surface elevations, 3 km above the confluence with the Ohio River, modeled slope error < 5% for low, medium, and high flow conditions.

Given in Appendix F, Figures F1 through F16 highlight various aspects of water surface elevation analysis. These include elevation model equations for the mainstem spanning minimum stages experienced during the study through the approximate 100-year flood of 1997. Elevation and slope models can exhibit deviation from true slope along short distances, where the constant slope models per flow condition should not be applied within or across single riffle-pool configurations. It was determined that for most stations assumptions for uniform flow prevailed across non-flood conditions. This indicated slope was constant with respect to the sediment bed and changes in water depth. For uniform flow, momentum and estimation of flowrates are more easily addressed, for example in use of Manning's equation in stage-discharge relations for fully rough hydraulic boundaries (Vanoni, 1977; Bedient et al, 1992; Yang, 1996).

C.4.2.1 Flood Crest Analysis

Extreme high flow conditions were evaluated under four scenarios including a) maximum flow measured during routine weekly sampling, b) bankfull flow, c) June 1996 flood crest, and d) March 1997 flood crest. Shown in Figures C1, C2, and C3, gage data for the floods in June 1996 and March 1997 were based on direct measurement of actual surface levels or recorded flood marks. In some latter cases, photo evidence, field notes, and analysis were needed in conjunction with site survey data to determine relative crest stage to the measuring point control. Review of flood data sets indicated crest levels for natural discharge to be fairly accurate (± 0.03 m to 0.3 m) for most sites collected, with larger errors possible at TS17. Uncertainty in natural crest stage increased at the creek mouth where modest non-uniform flow conditions can occur in conjunction with an Ohio River backwater effect that usually increases stage above levels experienced during natural crest.

More accurate then the 1997 data set, flood marks were directly measured after the flood of June 10, 1996 (~5 cm rainfall) where hydrographs at LCS9 and LCS5B also showed complex but similar shape to an approximate 100-year storm experienced on March 1, 1997 (~15 cm rainfall). Both storms expressed rainfall in two large pulses separated by 16 to 20 hours. Exceptions to this description are that the larger 1997 flood began under low flow conditions where the 1996 flood began under high flow conditions. Secondly, representing the northernmost headwaters and Athens County, station LCS1 missed a second wave of rain in the 1997 flood where it concentrated mostly in Meigs County. Wave celerity in the June 1996 event measured with hydrograph data at stations LCS5B and LCS9 was 4.3 km/hr. Based on Manning's equation and characterization of the Leading Creek lower mainstem as wide rectangular or parabolic, water velocity for the 21 km associated stretch would be expected to be 67% of wave celerity or 2.8 km/hr (Linsley, et al, 1978). Wave celerity was used in part to interpret crest stage in the 1997 flood to distinguish between backwater and natural discharge in Figure C3 where CGS1 and CGS2 were rendered inoperable.

C.4.2.2 Benchmark Survey Calibration Using Backwater Pool Data

To better understand backwater flow dynamics and to strengthen the accuracy of slope estimates for later use in extrapolating stage-discharge curves, backwater pools were also investigated. Backwater events and slopes are detailed in Figures F2, F4, and F5 in Appendix F. Solutions of flow and surface profiles are generally of the form incorporating pressure head, velocity head, and elevation head (Bedient et al, 1992). Analysis for gage relations under backwater influence were made for steady non-uniform flow conditions where $y_{depth} > y_{normal} > y_{critical}$, with respect to the associated control point (Bedient et al, 1992). Backwater analysis here represented the mild slope case with sub-critical flow examining for a single measuring event convergence of free surface slope $\rightarrow$ 0. Benchmark surveys for the staff gages were conducted in May 1997 using dual ground positioning (GPS) receivers or control traverse from USGS benchmarks, referenced to mean sea level (AMSL). Benchmark elevation survey error was reported at $\pm$ 0.03m for traverse control and $\pm$ 0.03 m to 0.3 m for sites using GPS, as indicated in Table E2 in Appendix E.

Calibration of slope-elevation models derived here was further refined using mouth pool data during various storm events occurring throughout the study period. In reference to the field benchmark survey, absolute error in slope with respect to sea level at each site was reduced at most stations near the mouth by an order of magnitude, from 0.3 m to roughly 0.03 m. Error in measured daily slope between mainstem stations below Little Leading Creek was thereafter dominated by an assumed stage measurement error of $\pm$2*[0.003 m to 0.006 m]. Due to vandalism, final TS17 errors in slope retain increased uncertainty ($\pm$ 0.03m) due to lack of additional independent verification of a change made to the onsite measuring point control during the study. Stated error in derived slopes for pools may be subject to natural fluctuations induced in the free surface (e.g. $S <> 0$) due to other factors, for example changes in water temperature. In reality, flood pools on this reach of the Ohio River are in constant transition due to human and natural factors. The assumption that the flooding pool surface slope is constant may be more valid throughout the day backwater crests than the assumption that $S = 0$. Slopes < 10^{-5} should not affect calibration results significantly which were otherwise limited by measurement error under these conditions. Selected pool events used in this analysis were characterized by relatively instantaneous “snapshots” of data collected typically across all sites in < 1 hr, measured in visually still waters, and all with free surface slopes < $3x10^{-6}$.

Dates with strong backwater influence inducing static pool conditions at the mouth of Leading Creek were identified graphically through gage relation fall analysis and continuous hydrograph data collected at station LCS9. Selected pool dates including 5/18/96, 5/19/96, 12/4/96, 3/4/97, 3/5/97, 3/12/97, 5/27/97, 1/13/98, 3/23/98, and 4/23/98 were used in determining benchmark survey offset corrections. The latter two dates were based on gage relations only. In summary, the backwater pool analysis illuminated the error in absolute GPS elevation control between the 5 study sites under the influence of the Ohio River, which exceeded reported tolerances by 50% at TS17. Benchmark corrections for head h_{ij}=0 was given by head offset values determined for LCS9 and TS17 during the extended flooding pools, shown in Figure E1 and Table E5 in Appendix E. The study assumed offset corrections for $h_{LCS7\text{-}LCS8} = h_{LCS8\text{-}LCS10} = 0$. Analysis indicated that LCS7 reached pool level only once in the study, during the March 97 flood. Insufficient data was available to determine if the modest offset measured for LCS7 during apparent pool conditions was induced by elevation or non-zero slope, where the latter was assumed. Benefits of controlling GPS absolute offset error in slope increases as slope $\rightarrow$ 0, near the creek mouth.

C.4.3 Stage-Discharge Relations

The method of velocity profiling followed the mid-section method of computing cross-sectional area and discharge for streams (Bureau of Reclamation, 1984; Rantz, 1982; Linsley, 1975). A stage-discharge curve allows one to convert stage or stream depth directly to flow rate; for instance from units of meters to units of cubic meters per second of water passing through a given cross-section of the stream. Data entry, analytical computation, and graphical computation software were developed and automated to allow for a consistent, accurate determination of area and flowrate for each profile. With flowrate a wide array of geomorphologic parameters including bankfull dimensions were also determined (Rosgen, 1996; Leopold et al, 1964). Multiple error tracking conventions were employed in programming. In all 135 profiles were conducted in the field at SGS, CGS, and other stations ranging from approximately 5 to 35 midsections per cross-section depending on the event, station, and flow condition. One event in late October 1996 characterized by low flow conditions included a sweep of 55 stations across the watershed where all but SGS stations utilized a five-point profile. For this event, tributary stations routinely monitored were surveyed in addition to 26 reaches located just below mainstem tributary confluence points. Added mainstem stations were selected based on primary tributary drainage area > 130 ha or 0.33% of total watershed drainage area. The event was instructional in assessing consistency of flow patterns across the entire watershed, sediment loading dynamics, and habitat quality at remote stations.

To improve flow measurement accuracy, depending on the profiling event, several depths of water (along a vertical) were also periodically measured (2/10ths, 6/10ths, and 8/10ths convention based on log law averaging). Measurements were repeated sometimes within a vertical at the same spot to serve as checks on the methods and field procedures developed for the project. For all measurements of velocity, the default 30-second averaging meter cycle was used for each recorded value. Repeated measurements at a single midsection location were first averaged to attain a final result (i.e. for each vertical taken in a midsection). Results inconsistent with an assumption of continuity in the velocity across the cross-section, or down the vertical, were also usually checked with additional, repeated measurements to verify proper meter operation. Anomalies during profiling were due to unpredictable aspects of turbulence associated with instream obstructions or substrate heterogeneity. Both wading and bridge measurements were conducted at all SGS sites. Under extreme flow conditions at bridge sites, to maintain consistent perpendicular referencing for the propeller, a deflection angle of the handheld aluminum pole was used along with a cosine adjustment of velocity. An example rating curve is given for station CGS1/LCS5B in Figure C4. Curves for all SGS stations in Leading Creek are presented in Appendix G in Figures G1 to G16.

C.4.3.1 Stage-Discharge Rating Curve Development

Shown in Figure C4, stage discharge development at each of the gage stations followed a course of separation of each station into curves describing upper and lower flow regimes, a sometimes advantageous practice (Linsley et al, 1978; Vanoni, 1977). Phenomena associated with discontinuities in stream rating can derive from hydrograph hysteresis as well as flow-variant particle size distributions associated with the transient nature of gravel and sand bed forms during higher flows (Diplas et al; 1994; Bedient et al, 1992; Yang, 1996; Vanoni, 1977). Discontinuity was not a severe issue for most sites in Leading Creek but choice of a single model form across multiple watersheds was desirable. The two-curve fitting process was amenable to easy solution of equations for low order polynomials, and was a distinct advantage in normalizing tasking associated with later data processing needed to generate flow estimates.

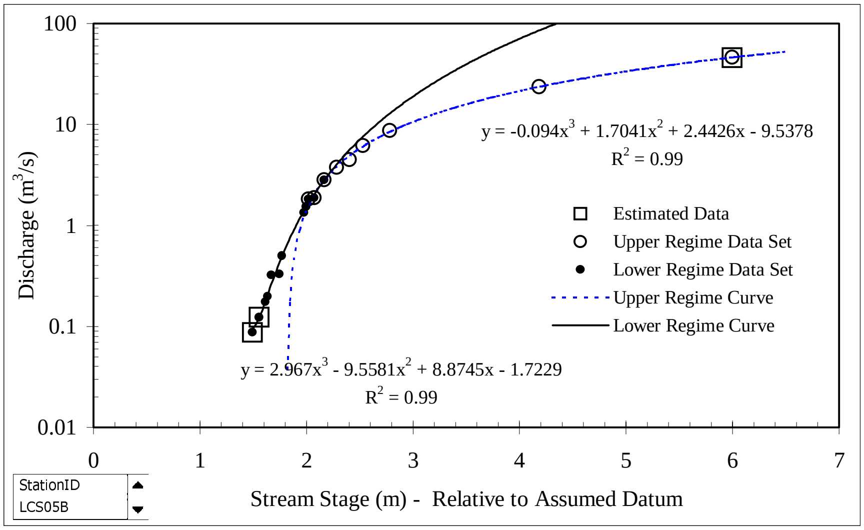

Figure C4 – Rating Curve for Leading Creek CGS1/LCS5B below Dexter, Ohio

Qualifications that follow are important to understanding the underlying methodology used in rating curve development across 16 gage sites. One aspect borne out by this ambitious undertaking was the realization that data processing and quality assurance tasking were greatly underestimated during experimental design. Other than measuring stage, velocity, and flow area, several factors are at play in watershed hydraulic assessments that can invalidate flow ratings and mislead interpretation if not accounted for. This becomes increasingly challenging when rating extreme low flow and bankfull flow is attempted. While benefiting end examination, multiple flood events can wreak havoc in rating streams with heavy sedimentation or that are otherwise under strong human influence. Equally difficult challenges exist when significant, poorly measured time-variant point sources are involved that invalidate assumptions of mass balance.

C.4.3.2 Baseflow Discharge

As a basis for beginning to examine station to station flow relations, Figure C5 presents September 1996 results representing the lowest flow survey conducted at all SGS sites during routine monitoring. The three-day survey of gage stations occurred without the presence of significant active mine discharges. Quarry (LCS1) and beaver/muskrat dam affected data (TS3A and LCS3) was omitted from the fitted curve. The survey results highlight three system features that had to be accounted for in modeling natural discharge rates. These include transient beaver and muskrat activity, and discharges from active underground mine and closed quarry operations. The baseflow profile event measured between 9/24/96 to 9/26/96 included insignificant daily total rainfalls of 0.03 cm, 0.04 cm, and 0.03 cm, respectively.

Stage data screening included review of weekly changes in head at sites, and between sites and adjacent sites. Matched rainfall data was also used relating totals for the event day and the previous day. This led to identification of some unexpected variation noted at LCS10 and LCS3. LCS10 variability was attributed to backwater effects. Strongly exhibited in the flow models for the study period, station LCS3 experienced periodic beaver activity and intercession by a local farmer. Figure F6 in Appendix F shows effects to slope observed. For brief periods in the fall typically, significant variation in surface slope and stage were seen at LCS3. As an example, severe beaver dam effects of a 0.15 m to 0.6 m dam build up during 10/15/97 through 11/15/97 were measured. Except for the fall 1996 low flow model data subject to a 0.1 m dam effect, stage-discharge data at LCS3 was based solely on non-dam conditions. Nominal equivalent uninfluenced data could not be recovered without use of the low flow model at LCS3. For this site lower flow ratings were calibrated by the baseflow flow equation in Figure C5. The final curve was corroborated by two independent predictions using flows at LCS2 and LCS4, where both agreed well with the curve estimate for LCS3 for those events.

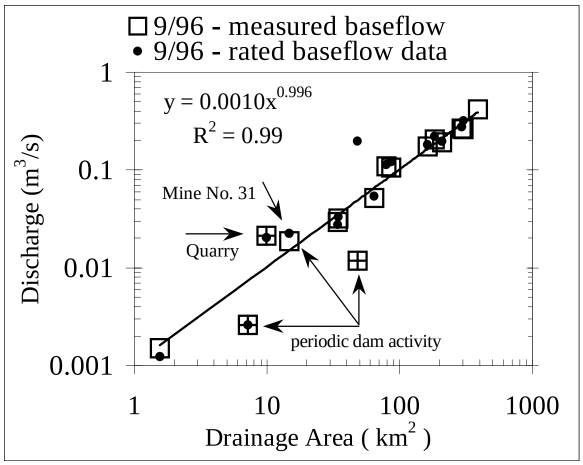

Figure C5 - Leading Creek Base Flow Survey. Regression significant ([P]<10^{-5}; α= 0.05).

Except for one period in fall 1997, the gage at station TS3A exhibited a relatively consistent dam influence effect throughout the study period attributed to a muskrat den, where no corrections were made. At medium flow conditions predominant backwater influences from the Ohio River explain departures observed at LCS10. Station LCS10 can be strongly affected by Ohio River backwater at low flow too, where drawdown below normal pool levels can also occur. During profiling attempts, flow was actually observed to cease for brief periods sometimes, evidenced by changes in movement of floating leaves on the water surface. This could be attributed to tributary inflows upstream on the Ohio River or hydroelectric dam activities. Daily lock activities though would not be expected to affect pool levels significantly. Low and medium flow LCS10 hydraulic data could only be used to assess backwater conditions.

C.4.3.3 Mine No. 31 Influences on Mainstem Slope and Flow Shutoff Events

Review of LCS5B and the confluence with Parker Run (TS11) indicated a subtle slope-stage-discharge relationship was associated with mine flow activity (Linsley et al, 1978). This affected station to station slope analysis and slope-stage-discharge relations developed at LCS5B. The fall between LCS6 and LCS5B induced changes in slope ranging less than 2 to 5% attributed to Mine 31. System-wide slope error analysis confirmed on the other hand that natural variation induced changes in station to station slope of up to 20 to 30%, shown in Figure F7 in Appendix F. Modest non-uniform (true station to station) slope characteristics resulting in average errors of up to 20% at lower flows and up to 30% for flood conditions were associated with some mainstem sites. Non-uniform expressions of channel characteristics were typically most pronounced in transition from extreme low flow conditions to medium flow, usually representing the initial 0.6 m to 1 m change in stage. Depending on flow condition, there was a natural rise in slope from 0.0005 to 0.0006 between LCS4 and LCS5B for example, and a subsequent decrease to 0.0005 again after LCS7. Slopes derived from tributary stations with respect to their confluence points along the mainstem showed consistently uniform flow characteristics across all flow regimes. Due to greater initial slopes defined by topographic relief, this was also the case for headwater mainstem areas.

A stage-discharge data point collected at CGS1/LCS5B on 4/25/97 was completed just prior to expression of an associated mine flow shut-off event. Review of other data confirmed the data point was developed for the nominally assumed condition of constant-effect discharge occurring at Mine No 31 ranging 0.17 to 0.27 m^3/s. The only exception was the low flow model rating point collected 9/24/96 during an extended shutoff, where final stage used in rating the site was adjusted upwards 45 mm to represent nominal conditions. Reported by SOCCO personnel, weekday head at the Mine Nos. 2 and 31 effluent discharge locations were recorded. Treated mine wastewater discharged from the Mine No. 31 process treated wastewater reservoir was normally adjusted every few days. Reported daily mine flowrates were based on Manning's equation, an assumed roughness coefficient for discharge pipes, and a single water depth reading. Weekend discharge rates, not required by permits, were in most cases based on the last recorded value for the previous week. Unpredictable variation in mine discharges day to day could be observed occasionally, particularly noting weekends where mine flowrates sometimes increased.

A well-defined relationship in head changes at LCS5B due to No. 31 flow shut-off was easily discerned from the continuous record, following a decay curve lasting about 3 hours. To correct for infrequent mine flow shut off and its impact to station data collected at LCS5B, the continuous station record at LCS5B was simulated to recover the nominal fall induced by shut-off, where specific magnitudes were observable for each event. Thus the record at LCS5B reflects assumptions that nominal Mine No. 31 flow occurred throughout the study period. Correction of continuous and spot measurement head data collected at LCS5B during severe shut-off conditions, applied for low to medium flow conditions only, should not impact slope elevation models. Above medium flow levels at LCS5B there was minimal expected effect from active Mine No. 31. This was based on typical rises in stage induced at LCS5B by the mine discharge ranging 30 mm to 75 mm. Worst case head changes at LCS5B might be characterized by a change of up to 150 mm at maximum Mine No. 31 discharge rates (0.4 m^3/s) and natural minimum flow conditions. This methodology could not verify that full shut-off occurred, only that significant changes occurred for periods of reported daily average flow = 0.

C.4.3.4 Quarry Effects at Low Flow

The low-flow data point for headwater station LCS1 was excluded from derivation of the baseflow model shown in Figure C5. Based on weekly head readings, data at LCS1 and LCS2 indicated sometimes constant or decreasing flow experienced downstream. Significant differences on the mainstem above and below the quarry discharge were detected in both flow and water quality beginning in late spring. Of some advantage to AMD inflows nearer to the creek mouth, the limestone quarry appeared to increase its influence upon the upper mainstem during drier conditions in summer and fall. Using paired data at the two upstream tributaries TS1 and TS2, monthly average pH increased 0.4 units passing below the quarry at LCS1, where it continued to drop downstream. The null hypothesis was rejected for H_a: Upstream pH <> LCS1 pH during low-flow conditions (n=9; [P = 0.0056]; α = 0.05, 1-β = 0.91). H_o was also rejected pooling data for all flow conditions (n=18; [P < 10^{-6}]; α = 0.05, 1-β = 1), but was accepted for medium and high flows combined (n=9; [P = 0.25]; α = 0.05, 1-β = 0.20).

For 11 sampling events conducted during low flow conditions over the study period, specific conductivity averaged 0.47 mmhos/cm at TS1, 0.45 mmhos/cm at TS2, and 0.69 at station LCS1, respectively. During low flow conditions in June 1997, alkalinity measured below the quarry's confluence increased 133% on average, reaching 304 mg/L. The limestone quarry operation footprint was estimated at 45.9 ha based on digitization of USGS quadrangle data (scale 1:24000). Sampled mainstem headwater tributaries TS1 and TS2 together represent 29.3% of the LCS1 catchment, compared to the quarry operation footprint representing 4.2% of the LCS1 subshed. Remaining portions above LCS1 were mostly woodlands.

Using specific conductivity as a rough approximation of mass loading, data for the October 1996 sampling event predicted a flowrate 80% above background. Assuming the nearly identical results at TS1 and TS2 were representative, the calculated flowrate closely agreed with the discharge rate of 0.021 m^3/s measured at LCS1 during the September 1996 baseflow survey. Both events represented similar hydraulic conditions where the baseflow model predicted a flowrate of 0.0099 m^3/s, 114% above background. A similar mass balance on calcium indicated a predicted flowrate in October 1996 113% above background. The data support the validity of baseflow survey and model results and indicate the quarry has a significant impact upon both flow and water quality. This analysis allowed distinction of natural flow rates and explained LCS1 flowrate behavior observed during the two-year stage sampling program. The closed quarry discharge is regulated by NPDES but flowrate data was not readily available.

C.4.3.5 Estimated Stage-Discharge Rating Data

Estimated data points usually included at least one low flow point depending on distance of the lowest directly measured data to minimum stage recorded. Ratings also usually incorporated a bankfull estimate, or one exceeding bankfull. The latter used cross-section data available from bridge profiles taken from the maximum directly measured flow rating profile. Wading cross-sections were also used to generate a flow value based on estimates of bankfull flow area and associated hydraulic radius. The latter did not typically reconcile with expected trends at most sites and were discarded. This was attributed to the difficulty in estimating (true) bankfull flow area and hydraulic radius from wading cross-sections measured under conditions representing only 20% of bankfull conditions, where bank heights were visually estimated. In comparison, bridgework at many sites included a directly measured rating point close to or exceeding bankfull conditions.

In curve fitting stage discharge relations a basic approach of using two 2nd or 3rd order polynomials was employed. These were fit to field data that included estimates discussed for extrapolation to extreme low and high flow conditions. When possible, 2nd order equations were used for all equations but particularly for the upper regime equation to better ensure that flood flows were not overestimated. The resultant lower and upper regime curves were than evaluated to determine an intersecting transition match point. A zero point was employed to ensure that nonsensical relations outside the domain of raw input data were not realized when assigning flows from the head measurement data sets for each site. This prevented assignment of negative flow and avoided problems with local minima in 3rd order equations. Zero and match points are presented in Table G1 in Appendix G for all rating curves. Rating curves developed for CGS1LCS5B and CGS2/LCS9 were based on 16 directly measured data points each. These two stations expressed only minimal hysteresis with a mixture of rising, constant and falling stages measured. Other stations had fewer rating points and therefore uncertainty in rating curve accuracy is increased.

C.4.3.5.1 Low Flow Estimates

For each site, the lowest stage rating point collected in September 1996 was used to extrapolate estimates to lower stage levels by first deriving hydraulic radius and flow areas using detailed cross-section data from those events. The only exception was TS8B that had a lower rating point directly measured in June 1997. For the September 1996 low flow data, cross-sections were all measured on subsection intervals ranging 0.15m to 0.3 m or less. Differential stage drops used for estimated data were simulated together across all sites based on two head measurement data sets representing the lowest encountered water levels for the study, occurring on 7/9/96 and 10/8/97. The earlier date represented lowest flows measured at tributary and upper mainstem stations, though dryer conditions may have occurred on some tributaries. The October 1997 period represented lowest flows encountered on the lower mainstem when the Mine No 31 discharge went offline temporarily. Active mine wastewater discharges were shutdown periodically for fish electroshocking surveys due to specific conductivity at low flow > 5 mmhos/cm.

Analysis of estimated discharge ratings for minimum stage indicated little difference between approaches reviewed. These included: a) solving for low flow bed roughness assuming minimum slope from slope-elevation models previously derived; b) solving for low flow slope and estimating roughness from either D_{90} calculated from pebble data or visual observations of channel morphology (Yang, 1996); or c) using Manning's equation to solve for a ratio factor incorporating slope and roughness. Free surface slopes realized instream during measured minimum low flow conditions dropped 1 to 2 orders of magnitude below values derived from slope-elevation (station to station) models. Of techniques outlined, the latter two were the most physically based and the last was used here for extrapolating lower portions of rating curves. Results were consistent with significant anticipated departures in constant slope within a given step-pool configuration. Where feasible, wading cross-sections were established in regions of transition from one bar to the next.

An exception to developing a continuous rating curve down to minimum stage at each site was preempted by dam activity. Zero flow was modeled at LCS3 and TS3A for extreme low stages based simply on standard approaches taken for all sites. This was believed to be consistent with direct measurements that showed markedly depressed unit area discharge rates at these two sites in September 1996, and as shown by the baseflow flow model in Figure C5. Thus pool levels decreasing up to 50 to 75 mm were monitored but zero flow was assigned below the lowest practically determined flowrate at these two sites. Consistent with field reports, modeling at AML impacted sites TS16 and TS8B showed no flow on selected events, particularly in fall 1997 when drought conditions were encountered.

C.4.3.5.2 High Flow Estimates

Estimates for high flow also used a slope-n factor derivation as in low flow. Here bankfull geometry and slopes were used in conjunction with solutions of Manning's n at the highest measured flow and associated measured slope for that measurement event. For higher flows, roughness estimated via Manning's equation in this way was usually always somewhat greater then calculations using measured D_{90}'s, where available. The D_{90} approach to estimating roughness at or above bankfull stage thus resulted in abnormally high estimated flows. The differences appeared to decrease as highest measured stage increased to near bankfull condition. Higher flow condition slopes calculated via the slope-elevation model likely see convergence towards an increasing true reach slope at the measured cross-section, perhaps explaining this phenomena. Because slope was well characterized at these levels, effective true bed-bank resistance appeared to be increasing at near or above bankfull conditions. The comparatively increased resistance calculated near or above bankfull stage was attributed to bank and flood plain features where channel economies associated with mobile pavement and scour, which result in reduced bed resistance, had already been realized. These observations were based on an assumption of expected continuity in the upper regime curve.

Generated during instream surveys, qualitative bed roughness estimated from table values compared well to values derived from D_{90} calculated on the mainstem. Some station errors were still high though and may be unacceptable for certain uses (Linsley et al, 1978). Error between visually estimated n and pebble count derived n for 17 cross-sections averaged 1.0% (σ = 14.2%). This excluded the two uppermost headwater sites with assigned table values of 0.035, which overestimated calculated roughness by 31% and 50%, respectively. This analysis makes use of biased estimators of D_{90} based on particle diameters >10 to 15mm (Fripp et al, 1993). An exception to high flow estimation approaches outlined here was that bankfull and near bankfull estimates for stations LCS8, LCS10, TS16, and TS17 were based on Leopold's bankfull proportion equation and well described rating data at station LCS9 (Leopold et al, 1964).

C.4.3.6 Stage Datum Changes and Backwater Effects

The flood of 1997 induced changes in stage datum of 0.19m and 0.07m at stations LCS8 and LCS2, respectively. The shifts were identified across two years of data via graphical analysis. Correction offsets were calculated by selecting stable low flow periods from each year comprised of 8 weeks of head data each. Natural annual differences in baseflow discharge were accounted for at unaffected station LCS4, determined by consistency of hydrograph patterns. Offsets to normal datum due to beaver and muskrat activity at LCS3 and TS3A where evaluated for specific periods in late summer and fall. Datum changes were evaluated using weekly head measurement data and graphical analysis. Infrequent beaver activity noted at TS11 receiving Mine No. 31 effluents was not corrected. This was due to an inability to determine natural stream conditions surrounding these events of which only one two-week period could actually be identified as probably affected.

The LCS7 high flow rating data point measured on 3/3/97 was influenced by backwater. Effective, increased flowrate at the measured stage was adjusted accordingly for use in stage discharge curve development. Slope (LCS7-LCS8) measured during the post flood crest profile event was at least 0.0001 at 173.5m above sea level with nominal slope of 0.00051. A fitted backwater curve using sites above and below LCS7 was developed and a station-event slope was determined to be 0.00022 ± 5%. An assumption testing backwater condition at station LCS7 on 5/19/96 was not supported by stage discharge data measured the same day, with normal slope estimated. For the latter lower mainstem backwater event, a stable pool profile on the mainstem reached somewhere upstream of LCS8 to elevation 169.6 m. The potentially affected rating point at LCS7 was collected at an elevation of 171.0 m. For the two events, Manning's equation was successfully used in ratio form to estimate effective slope or flow, respectively. Similar analysis concluded that subtle, variable backwater conditions prevailed at station LCS9 during profile events on 6/21/96, 2/28/97, and 5/25/97. Though instantaneous slope was unavailable, measurements leading up to or directly following these events indicated prevailing slopes of 3.0×10^{-5}, 3.9×10^{-5}, and 2.8×10^{-5}, respectively. Excluded from final analysis, only modest differences were observed in rating curves developed with and without affected data.

C.4.3.7 Velocity Meter Calibration and Deflection Angle Corrections

Velocity meter calibration was determined by evaluating meter response over a measured course using a pier on a still pond. Walking speeds measured with a stopwatch were varied, relating the mechanical frequency of the propeller/rotor combination to the course distance. Three separate prop/rotor combinations were used throughout the study where continuing calibration for each assembly was maintained at least semi-annually, typically quarterly. An averaged curve was developed based on calibration data collected before and after a given set of profiles. The study used 5.1 cm propellers initially though 7.6 cm propellers where used for most work. The 7.6 cm props were found to be preferable due to improved calibration control and better responsiveness at low velocity. A calibration curve showing meter response for a 7.6 cm propeller on a measured course of 5.5 m using still pond measurements and varied walking speeds is given in Figure C6.

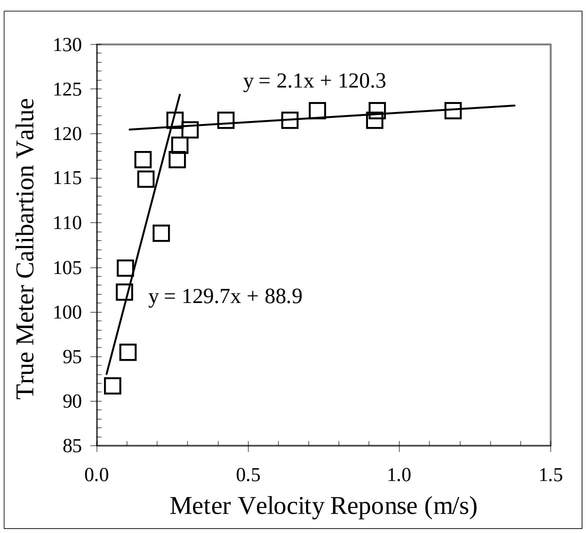

Figure C6 – Example Swoffer 2100 Velocity Meter Calibration Curve

Using 7.6 cm propellers for 89 cross-sections evaluated at the 16 gage sites, the average error adjustment in total flowrate was 4% (σ = 7.9%; maximum 25%). Averaged midsection adjustment across each profile event was on average 5.5% (σ = 5.6%; maximum 22%). Maximum midsection adjustment across each profile event was on average 19% (σ = 7.5%; maximum 37%). For these profile events, unadjusted width-depth averaged cross-section velocity ranged from 0.01

to 0.90 m/s with an average of 0.27 m/s (σ = 0.21). This data underscored the importance of controlling velocity meter calibration error in investigations spanning small and large subsheds. This tended to be most important at low flow when velocities were low and typical stage-discharge relations can magnify error between events and between sites.

Supplemental to meter calibration, under conditions with high velocities ranging above 0.75 to 1 m/s, depending on midsection depth, an induced angle of 2.5 to 30 degrees between the propeller and stream velocity vector was encountered. The propeller was mounted on a 2.5cm diameter aluminum pole with extensions reaching up to 7.3m. The observation depth fractions used in all work included 0.2, 0.6, and 0.8 depths referenced downward from the air-surface water interface. Pole deflection adjustment was made by dividing the measured velocity by the cosine of the deflection angle and was only needed for the latter two depth fractions. Of the 89 cross-sections referenced, 161 midsections or 6.5% exhibited deflection angles representing extreme high flow conditions. Maximum velocity measured was 1.35 m/s 0.97m below the water surface at an observation depth fraction of 0.6, at LCS1. Representing an exception to criteria typically used for determining the number of measured verticals per midsection, extreme flows were often conducted using a single 0.6 observation depth since the meter pole could not be controlled at the 0.8 depth and rate of change of stage was usually high. The number of midsections was also reduced to avoid the latter bias. For higher flow conditions, detailed cross-section depth data collected under calm conditions was used since bed depths could not be reliably sounded. This may represent bias in flow area calculations due to unaccounted for transient sediment scour or deposition.

The deflection angles were visually estimated in 2.5 to 5 degree increments. Under these conditions, the average non-zero angle observed was 10.5 degrees (σ = 7.0). Under extreme shear stress situations inducing high angles, bridge structures were used to advantage to pivot the pole to allow steady positioning. This angle correction method did not attempt to adjust for associated induced changes to observation depth and may have slightly underestimated true pitch since the pole itself experienced curvature and the propeller could not be visually observed at depth. The latter two bias effects would tend to offset each other to some degree. Observations under these conditions were physically demanding, dangerous, and usually required two to three people where one looked upstream to warn for debris floats.

C.4.4 Flow Regime Modeling

Empirical flow models based on the extensive head reading data set collected during the study period between 1996 and 1998 were developed in conjunction with stage-discharge rating curves. This provided descriptions for minimum, low, medium, high, bankfull and flood flow conditions at any site across Leading Creek. The model forms were fashioned on existing evidence that they should be strongly related to the form $Q \approx (A_d)^b$ where Q is flowrate and A_d is catchment drainage area (Leopold et al, 1964). The final flow models for the Leading Creek watershed, based on a non-linear function of drainage area, could not be improved upon by multi-variate non-linear modeling with surface slope. The statistical flow models as presented here are static with respect to flow regime, and do not account for many aspects of dynamic system behavior that are captured in more detailed hydrologic simulations driven by a forcing function based on precipitation, like HSPF (Donigian et al, 1995).

In order to develop a framework for flow regime modeling, the critical task was development of a meaningful set of threshold criteria that could be applied to the available data set to test viability of the concept. For practical purposes, this was strongly guided by a desire to obtain equivalent separation across low, medium, and high flow data sets with respect to log(Q). It was also desired to obtain a balanced separation with respect to the number of events populated in each group for the centralized gage used to represent the system. For somewhat similar reasons that the exponent b in the form equation is not constant, balanced separation cannot be obtained with a single criteria set across all areas. Smaller subsheds will be populated by more low flow data in random sampling programs then high flow data. This was not a severe limitation in the study of Leading Creek due to the large number of sampling events.

C.4.4.1 Flow Regime Separation Criteria

To develop appropriate geomorphologic-based flow regime separation criteria, inter-watershed comparisons were made with available data sets in the host Upper-Ohio River hydrologic unit. Final criteria arrived at are presented in Table C1. Starting with water-year 1991, mean annual run-off was evaluated over seven years at the closest USGS gage on Raccoon Creek (USGS Gage 03202000; drainage area = 1515 km^2). Normalized to drainage area, the mean annual run-off estimate for Leading Creek in Table C1 fell within the 95% confidence limit of Raccoon Creek's seven-year average of 1.14 MLD/km^2. An earlier comparative analysis of 56 Upper-Ohio River gage stations with records of twenty years or more also provided an estimate of 1.5 to 2 for the expected ratio of maximum annual flow to mean annual flow. This ratio was found to be independent of drainage area that varied in the study from 15 to $1.5x10^5$ km^2 (Linsley et al, 1978). Of the study period examined for Raccoon Creek, the maximum to mean annual discharge ratio was 1.2. This indicated that the study period for Leading Creek during 1996 to 1998, and for several years prior, represented average conditions for the parent hydrologic unit, irrespective of extreme floods or drought experienced. Sedimentation dynamics over this period would be expected to remain far more sensitive to these aspects with 9 bankfull exceedences encountered at LCS5B, when 1 per 1.6 years might be expected (Rosgen, 1996).

Table C1 - Leading Creek Watershed Flow Regime Model Descriptions

Stage-Flow Condition	Watershed Stage-Flow Characterization Based on Continuous Gage CGS1/LCS5B	Q_{LCS5B} ($m^3/s/km^2$)	Q_{LCS5B} (MLD/km^2)
All flow	Status assigned to flow-invariant data	-	-
Minimum	Represents minimum flow for study period	$5.8x10^{-4}$	0.050
Drought	Evaporation dominates in subsheds <10 km^2	$<1.0x10^{-3}$	<0.088
Low	Primarily baseflow discharge	$<2.5x10^{-3}$	<0.22
Medium	Transitional flow mixing baseflow with runoff	$>2.5x10^{-3}$	>0.22
High	Upper regime flow dominated by runoff	$>1.1x10^{-2}$	>0.95
Annual mean	Mean annual daily flow for study period	$1.5x10^{-2}$	1.3
Bankfull	Defined by local geomorphologic conditions	0.14	12
Flood	Representing conditions of extreme flooding	0.42	36

A comparative flow condition analysis of the daily records at LCS5B and Raccoon Creek were also conducted for these periods. Low-medium and medium-high flow regime thresholds were equivalent to 40% and 70% percentile rank separations (X(p[n+1])), respectively. This was based on average daily flow data at LCS5B for the only full water-year available, 1996. Comparison with daily flow data at Raccoon Creek from 1991 to 1997 indicated selected thresholds for low, medium, and high flow conditions on Leading Creek would represent equivalent percentiles of 35% and 66%, respectively. This assumed a simple drainage area ratio between watersheds. This broader study period also incurred both moderate drought and periodic flooding. Long-term and short-term records indicated that selected thresholds resulted in balanced partitions that were meaningful with respect to mean annual daily flow.

Drought in Table C1 is the least well defined. The threshold represents 40% of baseflow and is intended to indicate summer-fall patterns observed during this study when, for small subsheds <10 km^2 without inundating sediments, flow transport rates began to be overtaken by evaporation rates, evidenced by significant increases in mineral salt concentrations. Bankfull and flow relations are highly non-linear with respect to drainage area, and values reported in Table C1 are specifically for CGS1/LCS5B to allow comparison across all flow regimes. The selection of terms in Table C1 is based on conditions encountered over annual cycles. With respect to strict definitions that might otherwise be applied to terms such as baseflow and runoff, crossover will occur related to seasonal conditions (e.g. winter versus summer).

In reference to common separation of precipitation flow pathways into groundwater flow, interflow, and overland flow, the medium flow condition was considered to encompass stream conditions dominated by interflow and modest overland runoff in drier weather (Linsley et al, 1978). High flows were believed to be representative of primarily runoff and interflow during wetter periods. Minimum flow represented strictly base flow conditions classifying groundwater discharge. Low flow was considered primarily base flow though events characterized by this condition may also include interflow effects during dryer months accompanied by modest precipitation. In summary, daily average low flow on Leading Creek characterized in this study represented half the study period, 2 of every 5 days in 1996, and in general represented daily conditions when Q_{daily} < 16% of mean annual flow. Of 121 routine weekly stage measurement events analyzed for the study, 50 were classified as low flow, and 33 events were categorized as medium flow, and 38 as high flow conditions, respectively.

C.4.4.2 Leading Creek Flow Regime Models

Flowrate models using threshold criteria in Table C1 are presented in Figures C7a and C7b. Figure C7a represents threshold assignment based on rated CGS1/LC5B flowrate determined from a given event stage reading. Data in Figure C7b represents assignment of threshold criteria based on rated flows at each station. Flood of 1997 and bankfull data shown represent rated values based on unique stages for each parameter. Table C2 summarizes static flow regime equations for each condition. Equations were derived based on average data for each site for all events within a given flow condition assignment. Though difficult to see in the graphics, 95% confidence intervals are given for each site and low, medium, and high flow condition modeled. Station data below mines was adjusted to reflect only natural discharge conditions. Problems of accuracy in reported flowrates at Mine No. 31 were observed during data analysis. Therefore adjustment was accomplished for that discharge by estimating natural flow at TS11 from CGS1 instantaneous data, a relative area ratio, and an exponent value of b=1. The final value subtracted the non-natural component from rated TS11 data for each event from that station and those downstream. Mine No.2 adjustments to downstream mainstem stations were based directly on reported daily discharge rates. All regressions were significant ($P<10^{-5}$; $\alpha= 0.05$) in Figures C7a and C7b as shown in Table C2.

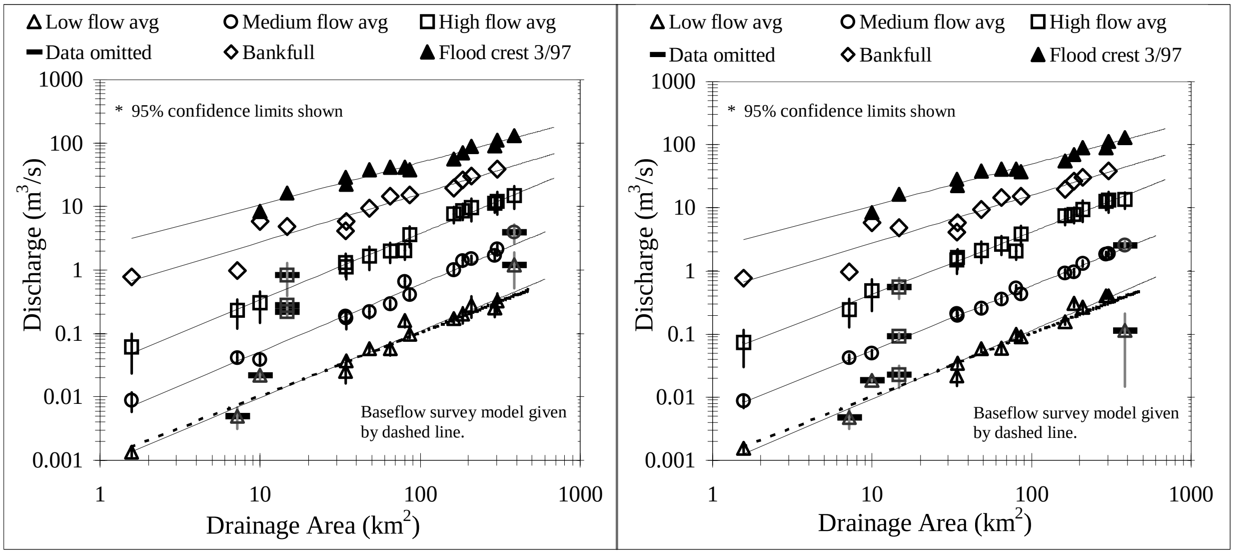

Figure C7a and C7b - Leading Creek Flow Condition Models With (a) Sampling Event Assignment by Flowrate at Central Gage CGS1; and (b) Site-Specific Flowrate Assignment

Table C2 - Leading Creek Watershed Flow Regime Models – Based on condition flow condition assignment by CGS1/LCS5B flowrate. Coefficients given in parenthesis represent site-specific flowrate assignment.

Stage-Flow Condition	$Q (m^3/s) = aA_d^b$; A_d = drainage area (km^2)		
	a	B	R^2
Baseflow	$1.0x10^{-3}$	1.00	0.99
Low	$8.6x10^{-4}$ ($7.7x10^{-4}$)	1.05 (1.08)	0.97 (0.98)
Medium	$4.5x10^{-3}$ ($5.3x10^{-3}$)	1.04 (1.02)	0.99 (1.00)
High	$3.1x10^{-2}$ ($4.4x10^{-2}$)	0.99 (0.98)	0.99 (0.99)
Bankfull	0.48	0.76	0.91
Flood	2.3	0.66	0.96

C.4.4.3 Data Sets Used In Flow Regime Models

Depicting natural flow, all data points were generated by rating curves and measured stage. Low, medium and high averaged flow condition models were based on weekly data and condition assignment by CGS1 daily average flowrate. Low flow data were omitted from curve fitting associated with the quarry at LCS1 and constantly dammed TS3A. For low, medium, and high flow conditions all TS11 data dominated by active Mine No. 31 discharges were omitted, as were Ohio River backwater effects at LCS10 for low and medium flow curves. For point sources and flow accounting used to adjust lower mainstem stations due to Mine No. 31 inputs, final models in Table C2 were insensitive to exponent changes between 0.996 and 1.08.

Excluding backwater attributable to the Ohio River, variation in slope at some stations for a few weekly stage measurement events were noted in graphical fall analysis and were attributed to rainfall and variable hydrograph position site to site. Backwater site-events were all excluded from development of flow condition statistics derived here. One weekly reading was identified as an outlier at LCS5B with an apparent unit error in tape reading (e.g. 1 ft), anomalous with continuous CGS1 data. Head data collected at sites when ice was present in January 1997 was also omitted from analysis and modeling. To avoid bias, beaver dam impacted site-event data for stations LCS3 and TS11 were not used in weekly flow condition models presented in Table C2 and Figures C7a and C7b. The same was true for a single period representing temporary dam removal at TS3A observed in fall 1997, and downstream mainstem data adjusted for Mine No. 31 inputs based on dam impacted site-events at TS11.

A single exception for assignment by CGS1/LCS5B was an anomaly at TS16 on 8/19/97 where the single data point increased average low flow by 430% (n = 47). It could not be determined if this was attributable to poor tape reading or due to localized rain effects. For CGS1 assignment, the omitted outlier exceeded the TS16's high flow average by 188% to 240% depending on use of raw average data or modeled average condition, respectively. Confirmed by anomalous flow data and interview notes, the first five weeks of data at TS16 was affected by changes in channel shape due to a local resident's attempts to dig out AML sediments with a backhoe 10 m above and below the bridge. This was done prior to station rating activity. The initial five weeks from late 3/26/97 to 5/2/96 were omitted from analysis giving time for the channel to stabilize after heavy flows. A minor dig-out effect was noted for spring of 1997 were a correctable 75 mm offset was observed for a period of 5 weeks until heavy flows were again realized. Due to TS16 bridge construction in December 1997, later data was excluded.

Flood stage in 1997 was not available at TS3A or TS16. Site cross-sections were inundated in wide flood plains and an accurate flood mark could not be developed. TS3A bankfull stage was confirmed to be 1.524 m and 1.527 m at two separate cross-sections. Station LCS1 was also thoroughly checked for accuracy of bankfull stage where no significant errors could be found. In agreement with data shown in Figure C7a, qualitative weather reports during the March 1997 flood indicated that station LCS1 in Athens County received less rainfall in the second wave of cells passing through the watershed. All other stations reside in Meigs County, comprising 96% of the watershed. Without the LCS1 1997 flood data point, the associated model was $Q = 3.0A_d^{0.62}$ ($R^2 = 0.96$). The estimated high flow data used in rating curves for LCS8, LCS10, TS16, and TS17 were not used for input into the bankfull equation solution shown in Figure C7a, repeated in Figure C7b. Because TS3A ratings represent constant dam influence, the station would be expected to depart significantly from watershed system flow models using drainage area, as seen in Figure C7a.

C.5 DISCUSSION

The partitioned hydrological framework for risk assessment developed here was intended to allow linkage of hydraulic data sets to key areas of watershed analysis, including water quality, toxicity, and land use analysis. The three areas are intimately related by water flow pathways and are often investigated in tandem in watershed investigations. A variety of methodologies can be considered in attempting to tie together hydrology, hydraulics, and metrics of ecology. In this study, investigation into quantitative "flow condition" or "flow regime" assessment led to development of a sound and simple approach to watershed characterization and flow model development. The hydrologic-hydraulic analysis was supported by time series data at continuous gage sites but relied heavily on statistics of staff gage head data comprised of 2700 head readings taken over two years at 16 staff gage sites, 1872 of which were collected on a randomized weekly basis. Other stage data described in Appendix D was used for independent calibration of CGS stations and determination of instantaneous flowrates for water quality sampling events. A portion of the non-weekly data was also comprised of serially correlated readings taken during flooding.

C.5.1 Quantitative Framework For Flow Regime Characterization

Instantaneous flowrate assessment for loading calculations, within a context of flow regime separation, is more entailed than described here. This work provides though the critically defining conceptual framework needed for segregation of these and other water quality data sets. Flow separation was based upon assumed static conditions given by threshold flowrate criteria with crossover interpretation to standard principles of hydraulic engineering and hydrologic simulation modeling. The framework defined can for example provide for well-defined segregation of pollutant loading rates, once determined instream, into meaningful hydraulic mechanisms that also determine concentration of mass. For example, contrasts between low and high flow regimes can be used to illuminate dilution effects upon (short-term) time-invariant loads and concentrations from active and abandoned underground mining sources. Similarly, high flow conditions may be used to test hypotheses related to the presumption of increased loading of pollutants associated with run-off and land surface disturbance. In the latter case an assumption *a priori* that pollutants are entrained from agricultural fields during conditions of increased shear stress upon the soil column is applicable (Bicknell et al, 1993; Linsley et al, 1978). Thus due to rainfall drops and increasing overland flow depths, quantified levels of aereal disturbance may lead to casual association that increased land surface disturbance increases unit area loading rates under high flow conditions.

Specific application of this flow regime modeling framework is reserved for more detailed evaluation in Chapter 6. Final notes are presented here that further outline assumptions, qualifications, and validation of this framework with respect to accepted geomorphologic principles.

C.5.1.1 Flow Regime Model Development

Flow models and gage relations developed here were inherently based on direct measurement of slope, head, velocity, bed roughness, and flow area along with direct measurement of channel shape. For low flow, the system stage-discharge curve set was also independently calibrated using a single directly measured baseflow event in September 1996 when no mine flows were present in the stream system, substantiating validity of adjustment procedures employed for significant point source discharges in the system. This analysis was based on conditions when significant backwater influence from the Ohio River and lock systems was not present. Assumption of mass balance was not used in development of stage discharge relations, other than as directly employed in the method of midsections. Mass balance was assumed in statistically based flow models presented here in making adjustments for minor groundwater diversion inputs reported at Mine No. 2 and directly measured inputs for more significant flows from the Mine No. 31 area. A final exception to this was use of mass balance in substantiating that quarry operations provided significantly higher flow rates during low flow conditions. Stage discharge ratings were developed independently from flow regime models. Finally, separation criteria were somewhat arbitrarily established, guided by objectives to provide balanced separation across all flows and available data.

Flow condition models developed for Leading Creek showed that drainage area can be successfully used across a wide range of flow regimes as a proportional factor in accurately predicting static concepts of flow conditions at unknown sites from similar statistics at a known site. Using a power function to model relationships between drainage area and flow, only modest differences were noted compared to strictly linear models (e.g. area exponent b = 1). This was true for all flow conditions except bankfull and flood. Exponents of drainage area fit for two independent low flow condition models in Leading Creek, using direct measurement of flow, were 1.06 for the October 1996 collection set used (n = 36; R^2 = 0.97), and 0.966 for the September 1996 baseflow set (n = 13), respectively. The latter 8-day survey experienced some minor rainfall effects at the end of the survey excluded from curve fitting along with non-quantifiable sites. The larger system wide sweep also indicated near-unit flow-area relations were consistent across all routinely monitored sites including gaged and ungaged tributaries, and the entire length of the mainstem.

C.5.1.2 Geomorphologic Based Flow-Area Relations

Indicated by Leopold, the power function exponent was expected to range between 0.65 and 0.80 for flood and bankfull discharges and 1.0 for mean annual flow in humid regions. Exponents relating discharge for less than bankfull flows increase in part due to water storage in stream valleys and more infrequent rain intensities that do not cover the entire catchment (Leopold et al, 1964). Shown in Figures C5, C7, and Table C2, exponents close to 1.0 can also be used to apply statistical separation of periodic data for flow regime concepts that extend from baseflow through low flows up through medium flow to high flow conditions as characterized in Table C1. This was successfully done using 121 measurements at 16 gages spanning large and small subsheds ranging 1.6 km^2 ha to 388 km^2 collected weekly across two years of stage monitoring. Modeling flow regime efforts were similarly successful using a single centralized gage CGS1/LCS5B on the mainstem, as well as assigning flow regime by site- specific flowrates listed in Table C1. Based on mean annual flow, threshold criteria developed here for separating data sets were meaningful for both short-term and anticipated long-term hydraulic records in Leading Creek. This was true for short-term comparisons within the study watershed Leading Creek and for long-term comparisons using unit-drainage area analysis and the gage record at Raccoon Creek located nearby in the same parent hydrologic unit. Here mean annual flow was closely characterized with the separation threshold for medium-high flows.

Comparing centralized versus site-specific assignment approaches, reduced variation in site-specific approaches was most significant for low and medium flows where impact from rain heterogeneity on certain days was dampened by splitting sites within certain events. This was consistent with expectations. Overall minimal differences between assignment approaches were attributed to large sample sizes available for most sites used in the centralized assignment scheme based on 38 high flow, 33 medium flow, and 50 low flow events, respectively. All models were based on an independent approach taken at each site to develop accurate stage-discharge rating curves based on a combination of direct measurements and additional estimated data points for extreme low flow and high flow events. The added estimated data were based on detailed measurements of channel geometry, and geomorphologic slope-roughness ratio relations based on Manning's equation. In some cases Leoplold's bankfull equation $Q \propto A_d^{0.75}$ was assumed. For the bankfull curve developed here, $Q \propto A_d^{0.76}$, estimated bankfull data was omitted from curve fitting. This indicated the watershed specific equation for Leading Creek was, on its own merits, a valid approximation of an average value anticipated. Conditions near and above bankfull become highly non-linear with respect to drainage area where increasing variation in bankfull discharge was seen in smaller subsheds < 50 km^2, shown in Figure C7.

Good corroboration was observed between independently derived geomorphologic relations developed in Leading Creek and those reported in literature. The importance of this is set out in four benefits. First, modeling efforts have provided a well balanced, quantifiable delineation of flow regimes for use in ecological risk assessments. This was based on flow models derived as near-linear functions of drainage area that attach meaningful separation with respect to common frameworks of baseflow, run-off, mean annual flow, and pollutant mass dilution concepts. Secondly, these curves can also likely be used as representative curves for nearby systems with similar morphogenetic traits (Leopold et al, 1964; Rosgen, 1996). Thirdly, and most importantly for development of mass pollutant loading rates, a high degree of confidence in flow ratings was independently established for gaged subsheds of Leading Creek. This last aspect also establishes, together with flow regime modeling results, a desirable capacity accurately extending gaged data on given days to ungaged watersheds of Leading Creek, with appropriate qualification of rainfall heterogeneity.

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C.5.1.3 Flow Modeling and the Rational Formula Method

One of the simplest and oldest methods for flow assessment and evaluation of mass loading is the rational formula Q_r = CIA, where I is rainfall rate, A is drainage area and C represents a run-off coefficient (Thomann et al, 1987). For agriculturally disturbed fields, unreclaimed AML, and active mines that have managed stormwater systems, the typical runoff coefficient C in Leading Creek would be expected to range 0.1 to 0.2 for Q_r in cfs, representing by and large rural subsheds. Slightly higher values of C for scattered small towns and villages might be expected. A rough approximation, the equation and flow regime modeling results infer that C, and I over extended periods of monitoring, would be expected to be relatively constant throughout Leading Creek. The rational formula supports methods here for statistical separation of instantaneous flowrate data to eventually assess differences in concentration, where runoff flow conditions were modeled as a simple function of drainage area.

Approaches for assessment of actual watershed loading rates utilized in this study are also supported by an accompanying concept of mean load per overflow event introduced by DiToro and described by Thomann (1987). In this method, the equation $W_r = c_{avg} * Q_r$ describes the relationship between mean concentration for the runoff event and the mean runoff flow, assuming generally that Q_r is independent of c_{avg}. As applied in flow regime modeling, this concept can be extended to statistics of multiple events sampled in a given flow condition. Low, medium, and high flow partitions will reduce temporally distributed dependency between c_{avg}

and Q_r, allowing for meaningful comparison of instantaneous single sample grab data collected under a wide variety of hydrologic conditions over two years.

C.5.1.4 Hydraulic Characteristics of the Dewatering Event

Reviewing the original AMD dewatering event by Meigs Mine No. 31 in 1993, the lowest flows of the decade recorded on Raccoon Creek also occurred in August to October, 1993 after heavier flows in July. Minimum, average and maximum flows for this period were 0.014 MLD/km^2, 0.042 MLD/km^2, and 0.29 MLD/km^2, respectively, with the maximum value falling just above the low-medium flow threshold condition defined in Table C1. This time frame roughly coincides with emergency dewatering activities conducted at Mine No 31. The data show that Leading Creek water quality was practically equivalent to untreated AMD discharged by the active mine for the majority of the dewatering period. Low flow levels in 1993 were repeated for the same periods in late summer and fall at Raccoon Creek in 1994 and 1995, where flows increased roughly 75% in 1996 for the late October period over years previous, returning to lower levels in 1997 and 1998. The 1993 flow levels in nearby Raccoon Creek were comparable to the same levels experienced in October 1997, the second lowest flow period measured in Leading Creek during this study. The data corroborates the accuracy of trends in drought and low flow discharge rates measured in Leading Creek.

C.5.2 Mainstem Slope and Geomorphologic Characteristics

Review of LCS5B and LCS6 based on slope calculations between LCS4 and LCS10 on 3/3/97 gave estimates similar to values developed at sites along the mainstem using flood marks, photos, and personal account of residents. Review of head data at LCS4 concluded consistently that the bridge deck relative crest was a fair estimate, and the GPS data for benchmark survey control appeared to be within tolerance. This was also indicated by independent review of 2.5-ft aerial contour data of the area collected by SOCCO. The independently gathered flood marks at LCS5B, LCS4 and LCS3 also generally confirmed that the benchmark at LCS4 is accurate. Anomalies in flood curve shape would increase severely if LCS4 low flow elevation was found to be more uniform. The benchmark GPS survey placing the LCS4 bridgedeck in the vicinity of 183 m AMSL appears to be accurate. Photo reviews with flood head data and elevation contour data show that the LCS4 stage datum would in fact tend to be overestimated, rather than underestimated.

Shown in Figure C3, the shallower slope manifested between LCS4 and Mud Fork (TS8) appear to confirm elongated pool structures found in this area along with several sand bed reaches exhibiting high silt/clay content with some reaches inundated by silt/clay. Lower washload measured at LCS5B on average compared to upstream stations corroborated this finding. The slope profiles for various stage-flow conditions represent a macrocosm of the common step-pool sequence found in a single reach, and intuitively relate to Schumm's characterization of the hillside-channel-delta morphology (Schumm, 1977). Seen in slope profiles with comparison to particle size data, the steepest portions of the system represent the hillside morphological element; with shallower water depths and slopes between 0.01 to 0.001.

The hillside zone is a water and sediment production zone characterized in Leading Creek by cobbled-gravel beds and run-riffle-pool sequences. The dip in slopes manifested at the hillside zone toe on Leading Creek mainstem at river kilometer 37 appear to be accurate, being corroborated by three independent survey data sources. Fed by the hillside, gravel beds and more diverse biota beginning in this area characterize a more ecologically productive channel zone. The channel is viewed as the morphological element serving as a transport conduit for water and sediment to the mouth (Schumm, 1977). Elongated pools in Leading Creek were less common in the hillside where beaver activity favored the hillside-channel transition areas. Not unexpected, particularly for relatively flat lying bedrock, this transition area also reflected significant changes in floodplain soil sequences. Interviews with residents in Dexter, Ohio indicated that aside from increased levels of fine sediment, existing channel characteristics in this region were for the most part natural and not induced by surface mining on Mud Fork or other activities upstream. Beaver activities were also noted to be consistently present there as far back as residents could remember (>60 years).

C.5.2.1 Particle Size Characteristics of Leading Creek

Pebble counts (Wolman, 1954) were completed on 21 cross-sections from headwaters areas downstream to LCS7. Representing biased estimators, average D_{50} and D_{90} were 24.8 mm (σ = 14.5) and 58.6 mm (σ = 34.6), respectively (Fripp et al, 1993). Three of four sites with $D_{90} \geq 110$ mm were believed to be augmented by human activity during bridge works or to create old wagon crossings. A few sites were different from those in initial habitat surveys discussed where Manning's n was estimated from book values (Yang, 1996). Figures C8a and C8b show variation in particle size fractions for fine material < 2 mm in gravel-bed and sands beds at routinely monitored tributary and mainstem sites. Fine material determined by hydrometer testing was classified as sand >0.075 mm, and silt/clay < 0.075 mm. Land use is contrasted between urbanization in small subsheds and strip mining and agricultural activities, respectively. Land use data in Figures C8a and C8b is described in detail in Chapter 3. A single distinction made here was that station LCS1 (Urban = 4%) directly below the quarry and two small urbanized headwaters subsheds was incorporated as urban dominated by lowering the previously used threshold from 5% to 3%. Urbanization > 5% was previously found to cause significant degradation to qualitative, ecological based habitat scores, including overall scores and particularly substrate subscores.

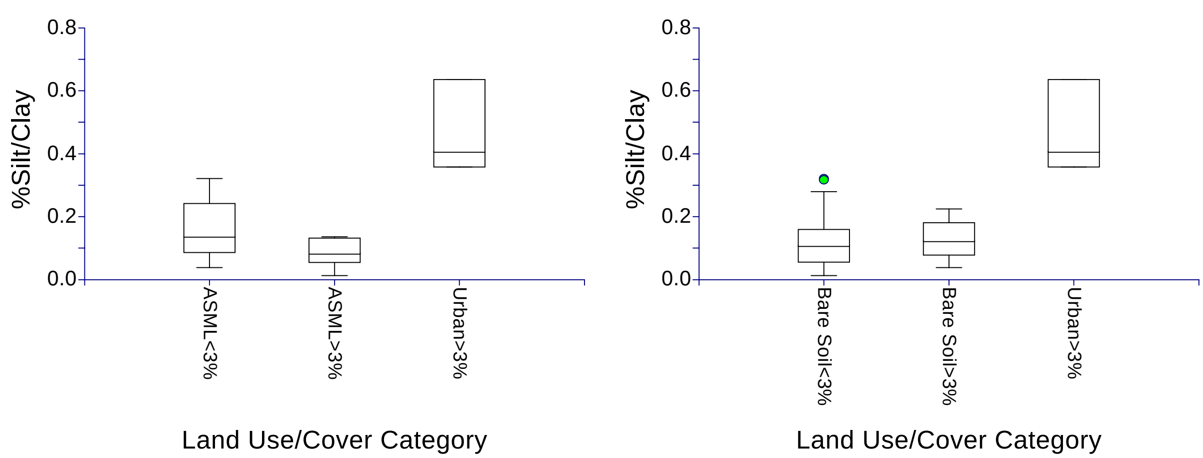


Figure C8a and C8b – Land Use/Cover Comparisons for Fine Material Particle Size Fractions Contrasting Urbanization and (a) Abandoned Strip Mines and (b) Agriculture

Parametric one-way ANOVA testing with multiple Tukey-Kramer testing was employed for fine particle fraction analysis with α_{MC} = 0.10, where tests met assumptions for normality and equal variance (Hintze, 1997, Zar, 1999). Results showed that significant differences in silt/clay content were observed in small heavily urbanized subsheds. This was first tested by evaluating strip mining land uses above and below 3% land surface area disturbance (n = 28, [P = $2x10^{-6}$]; α = 0.05, 1-β = 1.00). Significant differences in agricultural land uses were also tested in Leading Creek. Land use was defined by remotely sensed Landsat5 Bare Soil data taking during spring planting in June 1988 above and below 3% land surface area disturbance where reference site data was not available. (n = 27, [P = $3.1x10^{-5}$]; α = 0.05, 1-β = 1.00). For this latter test, skewness normality of residuals was borderline rejected with [P] = 0.0497. In multiple comparison testing, Urban>3% was significantly different from undistinguished differences in agricultural activities. LCS1 may be impacted by siltation from closed quarry operations, though this may be minimal since land surfaces have been reclaimed for greater than 10 years and stormwater run-off is controlled to degree. Two small agricultural subsheds and station TS11 below active Mine No. 31 with Bare Soil<3% exhibited low %silt/clay including TS9 and TS12 with of 32%, 28%, and 32%, respectively. Spearman-rank and Pearson correlation coefficients between %silt/clay and substrate habitat scores though were insignificant (α = 0.05; [P]=0.60, and [P]=0.078, respectively), were upper headwater tributaries showed unusually low scores compared to all sites including TS9, TS11, and TS12.

For urban contrasts with strip mining activities, multiple comparison testing with the reference site indicated similar results. Placing the single external reference watershed station (R)FCS1 (ASML=2.1%) within strip mining groups though indicated a significant difference existed between lower %sands at sites with ASML<3%, and higher %sands with ASML>3%. Fine sediment samples from gravel beds for toxicity testing of sediments are not volumetrically equivalent with respect to those beds, qualifying these comparisons. Importantly, gravel beds are not observable for the most part in cross-sections sampled for any subshed with ASML>3%. Thus data presented for those sites not only distinguishes relative fine material content, but for the most part absolute substrate characteristics, indicating the severity of ASML>3% on habitat. While a few of these sites retain gravel-bed hydraulic control at points downstream, all bed forms present would best be characterized as partially to completely inundated by sands (Diplas, 1992).

The mainstem of Leading Creek currently takes on a consistent sand bed nature below Lasher Run and other tributaries downstream feeding from the strip mine belt. The degree to which gravel-bed character would be expressed at lower streambed elevations without the presence of severe AML sedimentation is unknown. Personal accounts of long-term residents indicated gravel bed character in the channel earlier in the century did express closer to the mouth than is found today, but this could not be quantified. Changes in substrate were noted along with the disappearance of several deep pools previously found in hillside and channel regions. This latter distinction does not only implicate ASML but agriculture as well. The deeper pools and productive gravel zones no doubt contributed in part to Leading Creek's distinction for holding the Ohio State record earlier this century for a pike fish commonly known as muskie (*Esox masquinongy*), according to local residents anyway. Attempts were made to determine the veracity of this record but it could neither be confirmed nor disputed based on poor record keeping by the State, local newspapers, and fishing associations in earlier decades. This link to the past may represent the most well documented biotic benchmark available that Leading Creek just over 50 years ago was and is still today capable of expressing trophy habitat and biodiversity where little if any life now exists.

C.5.2.2 Effects of Sedimentation on Baseflow Discharge Rates

In comparison to expected trends, at extremely low flows a tendency for stream unit area discharge to fall-off on the lower mainstem below the mining belt was observed. While some of this could be attributed to poor ratings or subtle datum changes from 1996 to 1997 induced by the approximate 100-year flood, the effect was most extreme going from stations LCS8 to LCS9 to LCS10. Opposing the normal inclination to assign mass balance relationships under model calibration, these effects were not addressed in this way. Relationships developed for low flow were based on direct measurement and for extreme low flow were extrapolated using detailed measured hydraulic geometry and Manning's equation. The drop-off in unit drainage area discharge on the mainstem at extreme low flow may be attributed to massive inundation of sands in the mouth area. Significant portions of mainstem baseflow discharge during these drought periods are likely subterranean, moving through porous sands > 1m deep in some reaches below the Little Leading Creek confluence (Gallagher et al, 1999).

Lower than average low flow rates observed at station TS8B on Mud Fork were also attributed to excessive AML-related sand sedimentation and one of the watersheds few significant wetland areas which would be expected to reduce baseflow discharge due to upstream retention and increases in upper stem evaporation. The wetland is located in the upper portions of the watershed near station TS8A. Under drought experienced in later fall 1997, particularly in October, high flow inputs from Parker Run (TS11) above station LCS6 due to Mine 31 were not realized further downstream, presumably due to losses due to evaporation and previously noted tendencies for sediment inundation on the lower mainstem. Looking at weekly events, TS14 like TS8B, also showed a relatively consistent lower than normal discharge pattern during low flow conditions. This was apparently related to extreme ASML sedimentation in that tributary also. Meigs County utility workers reported that in the Town of Rutland evidence was found of > 1.5 meters of sand build-up on sanitary sewer line crossings in Little Leading Creek.

Unusually low flow at ASML>3% sites on the mainstem were also seen in the independently assessed flow condition models using weekly head data. During lowest flows and more so during drought, a pattern in weekly stage was observed where Mine 31 contributions to the mainstem at LCS6 were usually highly evident but marked decreases were observed by LCS7 with ASML<3%, indicating the primary loss mechanism here was evaporation. During the same periods, LCS8, LCS9 and LCS10 showed marked decreases in expected flow, attributed to increasing rates of evaporation and sedimentation. Under the most extreme drought period encountered in fall 1997, the lower mainstem stations LCS8 and LCS9 had discharge rates of 48% and 31% of predicted flows based on upper mainstem data and flow condition models. These two sites, characterized by inundated sand beds, represent severely impaired conditions > 20 years after most major strip mining activities have ceased. At the onset of strip mining decline induced by SMCRA in 1978, sedimentation in Little Leading Creek near Harrisonville, Ohio was documented to reach bridgedeck levels where up to six feet of headspace had returned by 1997 (personal communication with Mr. Byron Thompson, USDA Rural Abandoned Mine Land Program Director). These sands though still inundate lower reaches of Little Leading Creek near Rutland, Ohio, where significant portions of unreclaimed land still exist.

The impacted nature of sites TS17, TS14, and TS8B with heavy sedimentation made overall rating more difficult over time due to shifting sands. Cross-section shape was often observed to change throughout the lower reaches of these tributaries due to shifting sands, though flow area and velocity remained balanced. An example, middle bars (Yang, 1996) were common transient effects of ASML sedimentation in the mouth and larger, lower mainstem ASML tributaries such as TS17 and TS14. A consistent parabolic rating relationship was usually observed at these stations between flow area and discharge rate.

C.5.2.3 Elevated Low and Medium Flowrates In Thomas Fork

TS17 exhibited consistently greater medium and low flow condition flowrates with respect to other gage stations. Apparent low flow outlier behavior at TS17 exhibiting ~50% more flow than modeled rates for the rest of the watershed was determined to be a real and consistent effect of the system. Elevated discharge was observed during the baseflow survey and was observed more acutely during weekly monitoring. This could be in part due to the longer stream length and drawn out discharge, where often it took several days for the station to return to normal background flows which were still higher than normal. This was not however observed at TS14 in adjacent Little Leading Creek with similar geology and somewhat similar structure. TS17 station data was not adjusted where a 15 mm bed datum change may have occurred between 1996 and 1997 due to scouring of excessive sands. The latter was difficult to accurately verify and would have little impact on averaged flow data and models presented.

No anthropogenic activity is known to occur in the subshed that would explain the increased +0.05 MLD/km^2 levels observed. A thorough evaluation of head data sets, rating curves, and potential datum changes was conducted to assess potential error but no reasonable explanations could be found to explain the phenomena. At TS17, departures from watershed trends using site-specific flow condition assignment were still present but less severe, as seen in Figures C7a and C7b. A lower than expected high flow condition average and increasing average trend observed in transition to lower flow conditions could possibly be associated with the size of the Thomas Fork compared to characteristics of the reference station CGS1 used for assignment in Figure C7a. TS17 is the largest subshed tributary with a low flow slope of 0.0011 compared to station CGS1/LCS5B with slope 0.00059. Due to heavy mining activities, it represents excessive landscape disturbance and instream sedimentation that appears to increase temporary storage capacity and/or infiltration. The most remarkable distinction at TS17 was a high rate of abandoned underground mining with AUML=16%, which with its large size, may account for these differences. No conclusions could be drawn.

C.5.3 Velocity Calibration and Range of Non-Linear Responses

It is common practice for most field work applications to assume linear response over the full range of velocity for a Swoffer velocity meter. Shown in Figure C6, the popular Swoffer-2100 meter used here exhibited highly nonlinear response with respect to velocity magnitude. For water velocity < 0.2 m/s, frictional shaft resistance appears to become more significant. This may be characteristic of all prop/rotor assembly devices. The effect was corrected for using a graphically determined double line curve estimate with match point. The final calibration correction procedure employed an event specific function applied to each cross-section midsection, evaluated at each vertical point in a given midsection. This allowed for proper calibration over a wide range of velocities often encountered in moving from one bank to the opposite stream side, as well as up and down in each vertical. Above 0.2 m/s the prop/rotor assemblies become velocity-invariant.

Correcting calibration error is beneficial since the field calibration value assumed by the meter's signal processor might frequently be unrepresentative of actual measurement conditions. Once a calibration value is known for a given prop/rotor combination, setting the field meter as close to a consistently encountered true value is ideal. It would for instance be impractical to change meter calibration between midsections for a given cross-section. Because meter response relies on the mechanical frequency of the optical device (4 counts/revolution), post-measurement adjustment for small or large differences between field calibration and true calibration theoretically should not present concern. For each prop/rotor assembly the field value in this work was set to a representative true value for the linear range > 0.2m/s. For a meter set in this fashion, uncorrected errors in depth-width averaged flow calculations between midsections will cancel to some extent but may be significant if large percentages of total flow occur in low velocity midsections. Leading Creek profiling used mostly 7.6 cm propellers. For 89 associated cross-sections, the average error adjustment in total flowrate was 4% with a maximum of 25%. Maximum midsection adjustment across each profile event was on average 19% with a maximum of 37%. Results show that calibration of propeller type velocity meters in hydraulic investigations of small and large subsheds should be accounted for both in single-station rating curve development and inter-station analysis.

C.6 CONCLUSIONS

A variety of methodologies can be considered in attempting to tie together hydrology, hydraulics, and metrics of ecology. For intensive hydroecological risk assessment of a landscape radically altered by coal mining, this study found that drainage area and a set of well defined unit area criteria can be defined that together form a useful, meaningful framework for characterization of flow regimes. The partitioned hydrological framework for risk assessment developed here was intended to allow for linkage of hydraulic data sets to key areas of watershed analysis, including water quality, toxicity, and quantified land use/cover. Statistically based static flowrate power function expressions were developed for prediction across all portions of Leading Creek. Weekly static water level data collections over two years were used in assessing relative flow conditions from site to site. Initial derivation of absolute survey controlled surface slope-elevation models were found to be consistent with expectations and confirmed that assumptions for uniform flow were valid under most conditions for most sites.

Observations over a wide range of hydrologic and hydraulic conditions allowed meaningful separation of data and accurate description of drought through severe flood regimes, including Ohio River backwater effects. For all flow regimes encountered, expressions were nearly linear expect for bankfull and flood conditions. Exponents of 0.96 to 1.03 for low, medium, and high flow conditions were determined with lower expected values for bankfull (0.76) and extreme flood (0.72) realized. Results for the high flow condition model, with a criteria range similar to mean annual flow, were expectedly confirmed for a humid region. The noisiest, the bankfull model agreed exactly with average expectations based on hydraulic geometry (Linsley, et al 1978; Leopold et al; 1964). This work extended confident application of linear flowrate models of the form $Q = aA_d$ to low flow and medium flow regimes for the Leading Creek Watershed.

Benefits of flow regime modeling were set out by four conclusions. First, modeling efforts have provided a well balanced, quantifiable delineation of flow regimes for use in ecological risk assessments. This was based on flow models derived as near-linear functions of drainage area that attach meaningful separation with respect to common frameworks of baseflow, run-off, mean annual flow, and pollutant mass dilution concepts. Secondly, these curves can also likely be used as representative curves for nearby systems with similar morphogenetic traits. Thirdly, for development of mass pollutant loading rates, a high degree of confidence in flow ratings was independently established for gaged subsheds. This last aspect also established, together with flow regime modeling results, a capacity to accurately extend gaged data on a daily basis to ungaged watersheds of Leading Creek, given appropriate qualification of rainfall heterogeneity. Results of flow regime modeling described here were used throughout this dissertation to evaluate concurrent water quality monitoring data collected in Leading Creek. This work facilitated development of accurate quantitative relationships between water concentration data and risk assessment endpoints, specifically in relating unit area pollutant loading rates directly to quantifiable land use descriptions.

For all but one site, Thomas Fork (TS17), relatively consistent results were obtained in station to station relations for various flow regime models discussed. Thomas Fork with a drainage area of 79.6 km^2 consistently showed up to 50% higher flowrates during low and medium flows. The most remarkable distinction found in this subshed was its high rate of abandoned underground mining with AUML=16%, which together with its large size, may account for differences observed. Due to uniqueness of its size and magnitude of disturbance, conclusions though could not be drawn that directly contributed higher than normal unit area flowrates to AUML activity. In comparison to expected flowrate trends, at low flows and drought, a tendency for stream unit area discharge to fall-off in areas severely impacted by strip mining were otherwise observed, defined by ASML>3%, was observed. If similar effects were projected in Thomas Fork, noted increases would be higher. For urban land uses contrasted with strip mining and agricultural activities, multiple comparison testing indicated urban areas had significantly higher rates of siltation, whereas areas with ASML>3% had massive inundating levels of sand over gravel beds, and relatively higher proportions of sand materials >0.075 mm in total fractions < 2mm.

Finally, using propeller type velocity meters for flow rate determination, it was observed that meter response was highly non-linear with respect to magnitude of instream water velocities. A decreasing linear response below 0.2 m/s for 7.6 cm propellers was observed where above 0.2 m/s the prop/rotor assemblies tended to become invariant to changes in velocity. Using discrete transducers such as optical sensors that count propeller revolutions, a single meter calibration set-up can be corrected later in data analysis. This can be accomplished both along cross-sections and down vertical depths in the stream by accounting for specific velocities measured in each reading. For 89 associated cross-sections, the average error adjustment in total flowrate was 4% with a maximum single cross-section of 25%, and average midsection adjustment across each profile event was on average 19% with a maximum of 37%. Results show that calibration of propeller type velocity meters in hydraulic investigations of small and large subsheds should be accounted for both in single-station rating curve development and inter-station analysis.

C.7 LITERATURE CITED

See Chapter 2 for Literature Cited

Appendix D - Instantaneous Flowrate Modeling for Watershed-Scale Ecological Risk Assessment and Water Quality Monitoring Programs

Justin Eric Babendreier

Appendix D– Instantaneous Flowrate Modeling for Watershed-Scale Ecological Risk Assessment and Water Quality Monitoring Programs

Justin Eric Babendreier

(ABSTRACT)

A methodology was also presented to quantify instantaneous flowrates for water quality sampling events in ungaged subsheds on a daily basis. The analogy extended static geomorphologic flow regime models based on near-linear functions of drainage area to non-linear dynamic "event-day" models.

- For application in a watershed risk assessment context, instantaneous flowrate models based on daily time-steps can be constructed as power functions of drainage area for use in estimating flowrates at ungaged sites.
- Derived from gaged data in small and large subsheds, the geomorphologic-based models can be used to address heterogeneity in flowrates induced by spatial distribution in run-off.
- Based on geomorphologic approaches event-day models can also be used to assess rainfall heterogeneity and to strengthen overall quality assurance in head measurement programs for water quality sampling programs.
- Intended for grab samples collected during water quality monitoring, the approach allows accurate assessment of flow regime at ungaged sites and facilitates direct estimation of instantaneous flowrates for loading calculations and site-specific flow regime assignment.
- Point source average daily flowrates maintained in USEPA's NPDES Mor database can be inaccurate by up to 200%, and can seriously affect mass balance calibration approaches normally employed for hydrologic simulation of watershed flowrates and pollutant loads.

Appendix D – Instantaneous Flowrate Modeling for Watershed-Scale Ecological Risk Assessment and Water Quality Monitoring Programs

INTRODUCTION

The importance of a well-defined hydrologic-hydraulic framework in ecological risk assessment was further investigated here for application to watershed-scale studies of common anthropogenic stresses in Ohio. In the eastern portions of the state, these usually include some facet of coal mining. Underground mining or abandoned surface contour mining (i.e. strip mines or ASML) may be involved, and more typically both play roles. In the Leading Creek Watershed studied here, active deep underground mines and abandoned near-surface underground mines (AUML) are known to exhibit strong influences on water quality. The former act as two point source discharges and the latter along with ASML act as non-point source discharges to the stream system. Small subshed urbanization and widespread, variable influence of agricultural activities are also factors present. Non-mining aspects of human influence are themselves recognized as significant stresses in most watersheds across the country, and have been specifically identified in Ohio as sources of degradation to aquatic communities. Together with mining activities, all of these sources of stress must be carefully quantified and considered in holistic approaches seeking to associate land use and land cover management practices with water quality degradation. This task is an integral part of watershed risk assessment, if not the most important for land managers.

A critical aspect in engineering evaluations of water quality is the accurate determination of instream flowrates that lead to insight into the causes of its degradation. These insights are crucial to technically sound development of source pollution control approaches. Efficient success depends on proper identification and quantification of dominating mechanisms and processes. While flowrate estimation is an engineering problem spanning millennia, it is a relatively new task evolving with respect to applications in ecological risk assessment (Donigian et al, 1995). The work presented here is not in its own regard novel, though the scale of its application is to some degree, as is the ultimate focus of its attention to ecology. The methodology provides a simple approach of limited hydrologic simulation in ungaged subsheds that can be used across short time steps defining typical water quality grab sampling programs. It is built upon two ideas: 1) that direct measurements are available in nearby watersheds of similar size, and 2) fundamentally simple relationships that equate gaged subsheds to ungaged subsheds can be developed that are in principle applicable to the system studied. The novelty of this work lies in the construction and presentation of flow models used to successfully accomplish this task across a 2-year water quality monitoring program in Leading Creek. Necessary conditions to be successful are not identified. The work does though present an example of sufficient tasking that allowed accurate quantification of instantaneous flowrates in Leading Creek, and eventually, greatly enhanced exploitation of water quality data collected.

D.1.1 Parallels to Hydrologic Simulation

Hydrologic simulation models represent a distinct group of computer based watershed modeling tools available to the hydrologist and water resources engineer. Simulation models represent a class of flow models that can include address of groundwater, surface water, and other aspects of the hydrologic cycle like evapotranspiration (Fetter, 1994). Simulation models relate rainfall patterns to run-off patterns (Linsley, et al, 1975). Runoff is driven by rainfall intensity and duration, and will be a function of the watershed's size, slope, shape, storage, morphology, channel-types, soil types, and percent impervious land (Bedient, et al, 1992). Hydrologic simulation models have been classified for watersheds and cover a wide array of approaches and characteristics (Bedient, et al, 1992). The more complex models like HSPF, SWRRB and SWMM have an underlying physical basis, for example working on principles of continuity, and conservation of mass and momentum (Arnold et al, 1995; Donigian et al, 1995). These models all attempt to simulate hydrology of the system in some way, predicting among other things water flow on some incremental time step basis. A simulation is at some level just a series of static, discrete events. In hydrologic simulations, when dealing with several streams or reaches, a concept of flow routing must also be applied between these network elements. For watershed analysis, the parallels are normally drawn between use of lumped parameter versus distributed parameter models, event versus continuous models, and stochastic versus deterministic models (Bedient et al, 1992). In hydrologic simulations with HSPF, several reaches are often involved, and a concept of flow routing provides interconnectivity.

In context of work presented here, two facets of hydrologic simulation are always applied, regardless of their underlying structure and formulation. These entail elements of calibration, and then subsequently, application (Singh, 1995). In a hydrologic simulation driven by time-sequence rainfall data, discrete members of a subset of nodes in the network represent the calibration data set (i.e. gaged sites). These nodes may be simply extended through time in the application of the simulation. As well other nodes (i.e. ungaged sites) can be modeled at similar times that describe the calibration data set, and also extended through time. Here, one can envision both continuous calibration stations with time steps matching the forcing function (e.g. rainfall) and discrete (e.g. manual) stations where calibration data sampling rates are less than the forcing function time step.

From this framework, we can also envision how a water quality grab-sampling program for ecological risk assessment might be conducted. We first again generalize the problem by assuming water quality data was collected at the same times as flow calibration data was collected. We next assume water quality data is collected at all ungaged sites at these times. A first, simple question of interest arises as to how do we accurately predict flowrates at ungaged sites for the discrete set of times *t* matching our flow calibration data set. A second question posed and answered here, is do discrete simulations need to be driven directly by rainfall data to accurately predict those instantaneous flowrates at ungaged sites across a wide range of run-off patterns? Asked another way, can one forego complex hydrologic simulation modeling strategies based on a few points of calibration, and instead rely on a well selected, expanded set of calibration nodes to provide for productive analysis of water quality monitoring program data.

This discussion represents an over-simplification of baseline assessment, and does not extend to quantification of changes in land use/cover and effects of changes predicted instream. Nonetheless, the first problem of baseline assessment is a formidable one and of critical importance to successful ecological risk assessment. Certainly, if baseline assessment is not accomplished well, the extension to "what if" scenario modeling will be at best a wholly uncertain process. Even when done well, Melching (1995) has indicated that hydrologic simulation is itself a process with great uncertainty, where it may not be amenable to standard approaches of sensitivity analysis.

D.1.2 Instantaneous Flowrate Modeling Study Objectives

The utility of a watershed-scale water instantaneous flowrate modeling methodology was investigated in this work for application to ecological risk assessments. The methodology addressed aspects of hydrologic-based risk assessment by employing GIS based land use/cover analysis in conjunction with standard techniques for velocity and flow measurement. The objective was to provide a well-defined hydrologic-hydraulic water flowrate modeling framework for evaluating quantitative water quality data associated with agriculture, urbanization, surface mining, and active and abandoned underground coal mining land uses. Flowrate modeling was investigated in Leading Creek to facilitate an improved understanding of land use/cover impacts to instream sediment and water chemistry, toxicity, and biodiversity.

Modeling described here was intended to provide a sufficient level of discrimination in assessing hydraulically driven mechanisms. The umbrella ecological risk assessment of impacts from land uses focused on how land uses may be contributing to declines observed in biodiversity. Detailed watershed-scale hydraulic characterization for ecological risk assessments are not common on literature. This is particularly true for those that employ Geographical Information System (GIS) at fine scales of 1:24000, land use/cover analysis, and ecotoxicological based risk assessment. Hydrologic-hydraulic modeling reported here was developed as supporting tasks for multidisciplinary risk assessment of a large watershed (388 km^2) with those features in mind. The work tested an original hypothesis presented below

1) Using rated, continuous water level meters on the mainstem, and periodic staff readings from rated, manual gages dispersed across small and large subsheds ranging 1 to 400 km^2, periodic instantaneous flow rate can be successfully modeled in ungaged subsheds.

In the original experimental design, success implied development of an ability to predict site-specific flowrates that allowed consideration of water quality data along lines of low, medium, and high flow regimes, discussed in Appendix C. For overall design, an intensive approach to flow modeling and simulation was initially envisioned with application of a calibrated HSPF flow and contaminant fate/transport model (Donigian et al; 1995; Bicknell et al, 1993). The HSPF model was to be used to evaluate various changes to land use in Leading Creek. While the calibration data set presented here remains the same as intended for those discussions, study detailed here describes a highly simplified flow modeling approach that can be used for characterization of ungaged subsheds. The final flow models arrived at were based solely on drainage area and are not a function of quantitative rainfall. To support the validity of this approach, parallel characterizations of rainfall patterns for various monitoring events were conducted over low, medium and high flow stream conditions. A significant portion of this investigation was by necessity dedicated to analysis of uncertainty in National Pollutant Discharge Elimination System (NPDES) flowrate data. The data was collected in conjunction permit compliance activities. The external flow data is commonly employed as independent variables in simulation settings, and as shown here represents a potentially large source of error.

D.2 STUDY WATERSHEDS

Leading Creek is a 5^{th} order stream (scale: 1:24000) spanning primarily Meigs County, and portions of Athens, and Gallia Counties. Together, 11 mainstem stations and 19 tributary stations were selected for routine hydrologic, hydraulic, and ecological risk assessment monitoring. The monitoring network is detailed in Figure 3.1 in Chapter 3. Mainstem stations were located to provide complete longitudinal coverage of Leading Creek, and discrimination among potential sources of sediment or contaminant impacts to the stream system.

D.2.1 Leading Creek Mainstem and Tributary Stations

Ten mainstem sites and six of the tributary sites were included in stream stage monitoring indicated by designation as staff gage stations in Figure 3.1. Two continuous gage stations at mainstem sites LCS5B and LCS9 were continuously monitored for various parameters on a 5-minute time step. CGS data collection efforts provided continuous tracking of several water quality indicators describing instream environmental conditions throughout the study period. LCS5B is just above Parker Run's confluence and LCS9 is above Thomas Fork's, as shown in Figure 3.1. The two continuous gage stations LCS5B and LCS9 reflect significant differences in character between upstream and downstream mainstem landscapes. Agricultural activities dominate the upper mainstem while downstream reaches are dominated by mining.

Eighteen tributaries in the Leading Creek Watershed received some level of consistent sampling effort for most variables described here. These stations were initially chosen based on association with either (1) their known or suspected role in the contribution of point source pollutant loading to Leading Creek, or (2) their significance based on the size and nature of associated drainage area. The six tributaries selected for stream stage monitoring were chosen based on proximity to significant points on the mainstem of Leading Creek, proximity to known point sources and suspected nonpoint sources, or based on their significant representation of major tributary inflows. Parker Run (TS11) and Ogden Run (TS6) in Figure 3.1 receive active coal mining treatment plant effluents and assorted stormwater and sanitary discharges. Other NPDES sites relate to sanitary treatment (STP), water treatment (WTP), and quarry discharges.

D.3 METHODS AND MATERIALS

Data collection in Leading Creek and the reference watersheds included monitoring of: (1) Land use and land cover, (2) precipitation, stream stage, water velocity, and flow; (3) sediment and water quality; (4) benthic macroinvertebrate and stream habitat; and (5) toxicity through sediment, acute, and *in situ* testing programs. To assess chemical, physical, and biological interactions, stations in Figure 3.1 were examined over a wide range of drainage areas and stream orders (Minshall *et al*, 1985; Strahler, 1957; Vannote *et al*, 1980; USEPA, 1996c). Data collection methods specific to measurement of stage, water velocity, rainfall, flow and drainage basin analysis are detailed here, including methods used for water quality indicators.

D.3.1 Land Use and Land Cover

Spatial analysis of subsheds used several sources of spatial data including data hand-digitized from USGS quadrangle 7.5-minute map series (scale 1:24000), field data, and USGS hydrology geodata obtained from Ohio Division of Natural Resources (ODNR). Leading Creek subsheds were defined by digitizing hand drawn rain fall lines that also incorporated interpretation of the major influences of active drainage management associated with roadways. Initial conversion and processing of basemap ARC/INFO® exchange file formats for ODNR geodata was carried out using either ArcView® 3.0 and later versions or the Atlas® Import/Export program. Data describing locations of study sites, stream order, drainage area, stream length, and associated land use/cover characteristics were calculated using Atlas-GIS® Version 3.01 for Windows® and linked Microsoft Office® 97 spreadsheet-database programs. All statistical analysis was performed using the NCSS® 97 software package (Hintze, 1997) or routines in the Excel spreadsheet package. All raw and processed data was managed via linked ACSESS and DBASE III databases and pivot tables. The range of drainage areas describing 61 primary tributaries and associated subsheds in Figures A1 and A2 of Appendix A span 37 to 8038 ha each, where the smallest subshed routinely monitored for water quality was 124 ha, and the smallest gaged subshed was 157 ha. See Table A1 in Appendix A for specific drainage areas for each routinely monitored subshed and quantified geographic locations of cross-sections studied. All GIS data and area calculations were standardized to geographic coordinates using the North American Datum (NAD) of 1927 and retain horizontal positional accuracy ± 15m.

D.3.2 Continuous and Manual Gage Stations

A hydraulic network of 16 continuously or manually gaged sites was developed for Leading Creek. Hydraulic monitoring was accomplished using two continuous stream gages that split the mainstem, and 14 other manual gages placed in small and large tributaries. Equipment at each continuous gage station in Figure 3.1 included an ISCO Model-4230 bubbler level meter and Model-675 automatically tipping rainfall gage. Continuous monitoring with automated data storage allowed comparison of stream stage (± 0.025 cm), flow (± 10 to 20%; estimated), and rainfall data (± 0.025 cm). Flow data accuracy is site dependent and regime dependent. A sampling frequency of 5 minutes was used for all parameters. Approximately monthly maintenance visits were conducted to calibrate instream equipment, to check rain gages, and to download continuous data collected onto a portable AT&T 386 lab-top computer.

Each continuous gage station was powered by two 6V Trojan 120 amp-hour golf-cart batteries wired in series, allowing 3 to 6 months before recharged-replacement units were switched-out, and was typically done on a monthly basis. The continuous meters were housed in 1.5x1.5x1.5m wooden houses located on second terraces and all interconnections buried in 5 cm PVC conduit, except rainfall gage wiring which was buried at shallow depth using heavy duty commercial wiring with UV protected coatings. Rainfall gages were set in open areas 3 m off the ground with cover from wind provided by tree lines > one height away. The instream transducers for bubble lines and water quality sondes were securely mounted instream separate from each other. Conduit for sondes was exposed at the bank top, run downstream along the bank, and was secured off the sediment bed with anchoring. This allowed access for retrieval and calibrations. Sand build up at station CGS2/LCS9 needed to be periodically cleared.

Southern Ohio Coal Company (SOCCO) personnel collected weekly stage readings throughout Leading Creek. SOCCO personnel also routinely monitored daily rainfall at their Mine No.2 Coal Office at Point Rock, Ohio and the Mine No. 31 site in Figure 3.1, referred to here as No. 2 rainfall and No. 31 rainfall, respectively. Process treated wastewater flowrates from Mine No.31 and Mine No. 2 shown in Figure 3.1 were also routinely monitored on weekdays for NPDES compliance monitoring and reporting. Manual staff gage control points (inverted guides) were constructed of 2.5 cm diameter PVC tubes, approximately 0.5 to 1.5 m in length, and were rigidly attached to a bridge guardrail. Use was also made of a Solinst-100 electronic water level meter with audio signal and environmental probe or a steel tape measure together with line-of-sight reading to the tape end and water surface, both with similar accuracy (± 0.03 cm). Each top of PVC measurement was eventually transformed to a professionally surveyed vertical elevation datum. Surveys tied each gage to a known benchmark above mean sea level with elevation accuracy of ± 0.03 to 0.3 m. Absolute vertical control for staff gages was achieved using dual-unit geographical positioning systems or rod and level, and was later calibrated for lower mainstem stations using Ohio River backwater pools, reducing most absolute errors to ± 0.03. Prior to benchmark surveys, on-site vertical elevation control was established at start-up through two control points using an Pentax Pal - 5C autolevel and 13 ft wooden rod. As feasible, one spad each was placed on the bridge structure and one on a nearby mature tree or utility pole. Detailed survey control data is documented in Appendix C.

D.3.3 Channel Characteristics, Velocity, and Flow

Measured sediment properties, channel geometry, and observations in the streambed were used to characterize geomorphologic properties site to site. A combination of pebble counts, sieve analyses and hydrometer methods were selected for particle size analysis (Wolman, 1954; ASTM, 1990b, 1993; Gee et al, 1986). Limited sieve and hydrometer data were provided via routine sediment toxicity testing procedures establishing %sand, %silt, and %clay in samples. Stream stage height was calibrated to periodic velocity discharge measurements collected at each staff gage to allow for computation of stage-discharge rating curves (Bedient et al, 1992; Linsley et al, 1975; Yang, 1996). Stage-discharge ratings at each site were based on 5 to 16 data points collected approximately monthly at CGS sites and quarterly at other staff gage stations between April 1995 and June 1997. Each station rating included low flow data from system wide surveys conducted in September and October of 1996. One of the quarterly surveys was also conducted during a 100-year flood event, helping provide direct measurements across very low through high flow regimes. Added rating curve points were also considered and included a high-flow estimate derived from Leopold's bankfull equation $f \approx (\text{drainage area})^a$. For gravel bed reaches, supporting hydraulic analysis was evaluated, incorporating pebble count particle diameter (D_{90}), reach geometry, and Manning's equation (Vanoni, 1977; Yang, 1996; Leopold et al, 1964).

Velocity measurements at 1 to 3 vertical depths were taken with a propeller type Swoffer Model-2100 velocity meter using some 5.1 cm propellers, but mostly 7.6 cm diameter propellers, each continually calibrated with still pond measurements. Matched rotor-propeller sets were always maintained where two rotors were used. Wading and bridge cross-sections were established at each site to allow profiling during low to high flow conditions. The propeller mounted on a graduated (3 cm) aluminum pole 7.3 m in length, collapsible in 0.91m sections, and approximately 3 cm in diameter. Calibration of propeller/rotor combinations used were maintained throughout the study and followed procedures outlined in Appendix C. Velocity was calibrated after measurement based on event-specific, site-specific, mid-section-specific, and vertical depth-specific velocity magnitude measured.

A Keson nylon surveyor's tape was used for width measurements (61 m; graduated 3.1 mm). Reference stations were monitored for all parameters except stage-discharge rating activities. Flow computation followed the method of midsections (Bureau of Reclamation, 1984; Rantz, et al 1982). If encountered during higher flow, pole deflection angles of 5 to 30 degrees were estimated visually and corrected for later in data analysis. To maintain position instream, use of the bridgedeck and guardrails was often needed to brace the pole for currents > 0.6 to 0.75 m/s. Generally all length measurements were recorded in English units and converted to metric units during data processing. Through subsequent flow modeling efforts, flowrates at ungaged sites were determined as a function of drainage area using both a central gage with data from CGS1/LCS5B and site specific evaluations.

D.3.4 Water Quality Indicators

Water quality sampling was conducted at all mainstem, tributary, and reference sites shown in Figure 3.1. Monthly water column grab samples for pH, specific conductivity, and water temperature were collected and analyzed in the field, along with periodic monitoring for dissolved oxygen (DO). Water column samples used standard water and wastewater methods (APHA, 1992). Possibly not inclusive of all data collected, field meters included Corning Model-90 Checkmate multimeters, an Orion Model-122 conductivity meter, a YSI-55 oxygen meter, and a Fisher Accument Model-1003 pH meter. Specific conductivity was normalized to 25 ^{o}C (1.91% standard temperature coefficient; ± 1 umho/cm) and water temperature (± 0.01 ^{o}C). Portable pH probes were calibrated with pH 4, 7, and 10 lab-grade buffers and conductivity was calibrated with standardized KCl solutions using lab-grade deionized water (APHA, 1992).

Supplementing site-specific hydraulic data, equipment at each continuous gage station in Figure 3.1 included a YSI-600 water quality probe installed instream for easy retrieval. Continuous monitoring with automated data storage allowed comparison of temperature corrected pH (± 0.01 s.u.), specific conductivity normalized to 25 ^{o}C (1.91% temperature coefficient; ± 1 umho/cm), and water temperature (± 0.01 ^{o}C). During routine maintenance visits at CGS stations, conductivity was calibrated with fresh 0.005 N KCl solutions, and pH 4 and 7 buffers were used (APHA, 1992). Stage at gage sites was routinely monitored during water quality sampling events. Indicator data described here was useful in some instances for understanding flowrate dynamics from NPDES discharges associated with active mine operations and the closed quarry shown in Figure 3.1.

D.4 RESULTS

D.4.1 Continuous Stream Flow and Water Quality Indicators

Detailed descriptions and analysis associated with stage-discharge ratings for each gage site are described in Appendix C and Appendixes E and F. In Appendix G, stage-discharge rating curves used for work investigated here are provided for the 16 gage stations shown in

Figure 3.1. Figure D1 represents the duration of continuous gage monitoring activities at stations CGS1/LCS5B and CGS2/LCS9. Gaps in CGS records indicate backwater, malfunction or other data loss. March 1997 approximate 100-year flood peaks are not shown where CGS2 was disabled for this time period. Missing active mine daily data in Figure D1 indicates discharge rates were < 0.01 m^3/s. Figures 5.2a and 5.2b in Chapter 5 give summarized data for pH and conductivity across Leading Creek. Specific conductivity highlights differences between AMD impacted reaches with moderate levels, Mine Nos. 2 and 31 and lower mainstem sites with highest levels attributed to treated AMD, and decreasing levels from LCS1 in the upper mainstem attributed to discharges from a closed limestone quarry operation. Specific conductivity in Figure D1 also highlights differences between the upper and lower mainstem.

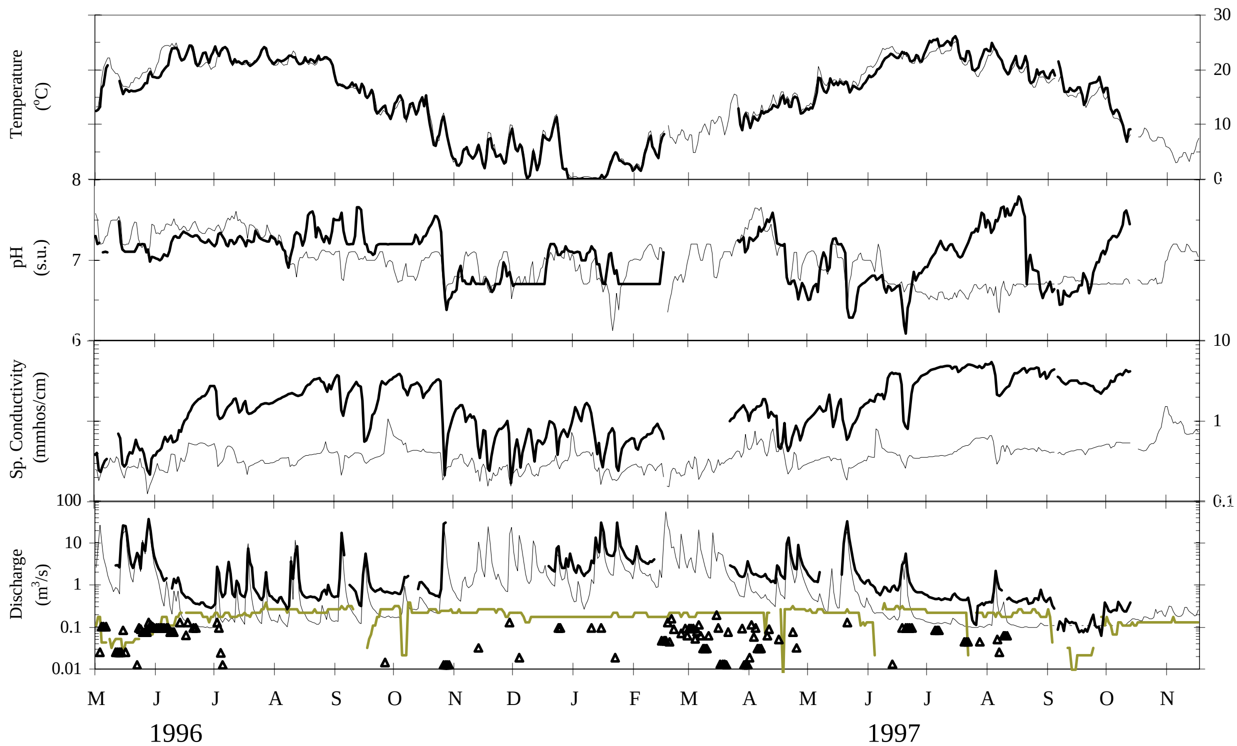


Figure D1 – Leading Creek Daily Average Data. Station CGS1/LCS5B given by light line, station CGS2/LCS9 by darker line; Mine No. 31 by dotted line; and Mine No. 2 by triangles.

In comparison to active underground mining, minor point source permitted National Pollutant Discharge Elimination (NPDES) discharges given by the WTP, STP in Figure 3.1 and sanitary wastewaters from the active mines represent insignificant loading of treated effluents and flow into the stream system. The active mining operations must meet categorical effluent limitations established for coal treatment (coal washing) plants. Coal preparation plant wastewater effluent is high in total dissolved solids (TDS) primarily composed of sulfate, sodium, and chloride where conductivity at SOCCO Mine No.31 typically ranges between 4000 to 6200 umhos/cm. During lower flow conditions the watershed can gain significant alkalinity from deep long wall mining mine operations, helping buffer AML-AMD entering the lower mainstem. The closed limestone quarry discharge is also a significant contributor of headwater alkalinity during low and medium flows. For most periods of flow, Mine No.2 wastewaters are conveyed to Mine No. 31 for treatment and storage before final discharge to Parker Run (TS11).

Following periods of increased groundwater infiltration after run-off, some process wastewaters are treated and discharged to Ogden Run (TS6). To assist data interpretation, flow, rainfall, and water quality data at SOCCO Mines No. 31 and No. 2 from January 1996 through July 1998 were assimilated into the watershed database. This data was helpful for early monitoring since construction and installation of continuous gage stations were not brought online until 5/14/96.

D.4.2 Manually Gaged Stream Flow

Shown in Figure D2, low, medium and high flow regime models are described for Leading Creek. These models were based on weekly stage measurements and daily assignment by CGS1 flowrate data collected between March 1996 and June 1998. Described in detail in Appendix C, threshold criteria separating various flow regimes were established to allow for a balanced analysis of flowrate data for both weekly sampling of gages and independent monthly sampling associated with water quality monitoring. Observations were taken over a wide range of hydrologic and hydraulic conditions and allowed meaningful separation of data and accurate description of drought through severe flooding. Stream slope analysis was an integral part of stage-discharge rating curve development at each site and allowed critical identification of Ohio River backwater effects at lower mainstem stations of Leading Creek. For low, medium, and high flow regimes encountered, expressions for flowrate were nearly linear with respect to drainage area, expect for bankfull and flood conditions with power function exponents of 0.76 and 0.66, respectively. In Figure D2, the medium-high flow separation criteria selected, was later shown to be closely analogous to mean annual flow. A critical design concept in flow regime model development was capturing an ability to relate discrete flow conditions to concepts of dilution, and baseflow and run-off that are often used to characterize watershed hydrology.

Range of flows for periodic stage, flow rating data, and water quality sampling are shown in Figure D2 for comparison. These provide an overview of the hydraulic and water quality sampling program and detail the relatively full coverage obtained at most sites during the program. This is an important facet of analysis and defines the validity of later works that use this data to infer effects of land use upon aquatic communities. Though difficult to translate into quantitative expressions of uncertainty in that analysis, data importantly show the wide ranges of flows were sampled. Also shown is the 1996 base flow survey model and linear models for mean annual flow, lowest flow measured at CGS1, and flow condition thresholds. Minimum flows of 0.001 m^3/s indicate flow rates were below practical quantification. Flood data shown represented an approximate 100-year recurrence interval according to stormwater management data provided by SOCCO which was assimilated as part of recent NPDES permit applications

D.4.3 Stage Error Analysis

D.4.3.1 Continuous Gage Stations CGS1 and CGS2

Table D1 indicates the overall stage error rates and % stage error rates for CGS1/LCS5B and CGS2/LCS9 station operations conducted over the study period. The analysis is based on post-calibrated station data and comparisons to independently collected weekly sampling data. Rates are also shown indicating data for station calibration events conducted on an approximate monthly basis. The calibration data was used to correct for drift offsets that occurred over time and was carefully collected with a metal tape each time. Calibration errors are nonzero after adjustment because of time delays that exist between the direct measurement and machine reset, and due to later data manipulation that proportionately adjusts for drift over the period between two calibration events. Drift can occur for example due to changes in elevation of the end of the bubble line caused by storm events. Negative error indicates meter data was lower than manually measured data.

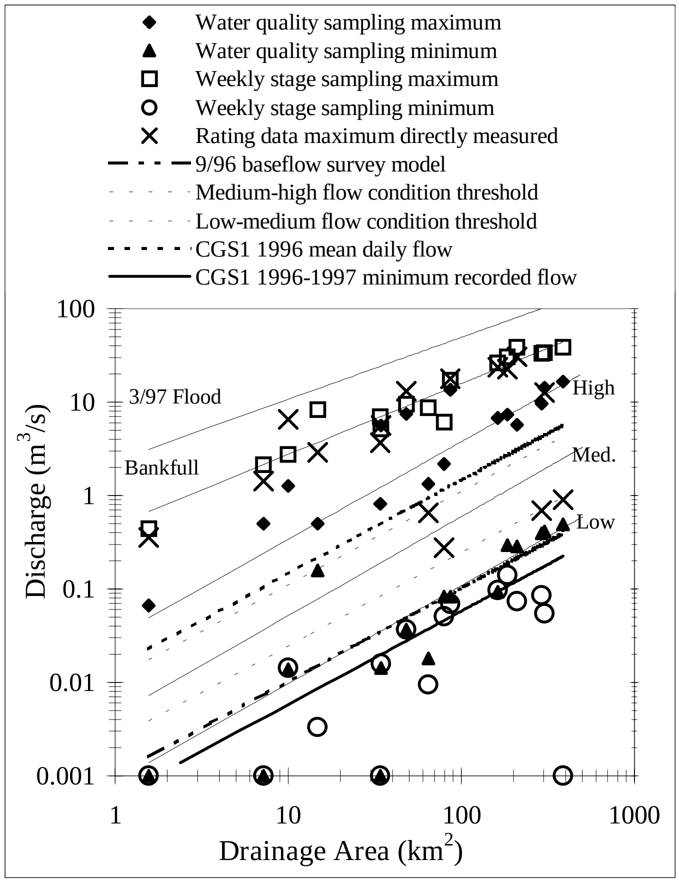


Figure D2 – Leading Creek Flow Regime Model and Hydraulic Monitoring Summary

Table D1 - Error Analysis of Stage at Continuous Gage Stations CGS1 and CGS2

Continuous Gage Station	Calibration Events		Weekly Events	
	Stage Error (mm)	%Stage Error	Stage Error (mm)	%Stage Error
CGS1/LCS5B	n = 34		n = 78	
Avg	0.44	0.02%	-4.0	-0.18%
σ	3.4	0.18%	22	0.95%
CGS2/LCS9	n = 42		n = 53	
Avg	-0.34	-0.02%	3.3	0.58%
σ	12	0.99%	60	2.7%

Continuing calibration of auto level meters was conducted approximately monthly or more frequently to correct for meter drift or channel variation due to sedimentation and scour. Average errors encountered in instantaneous head measurement data sets after calibration were ± 0.02%, shown in Table D1. Weekly measurement data set errors ranged consistently less than 1% of head except for a handful of selected events, some of which could not be explained. One event was determined to have the incorrect date assigned, offsetting manual and continuous records, while several other events appeared to exhibit clock offsets of 1 to 2 hours, where similar effects were often noted at both CGS gage sites when this occurred. In some cases the errors were possibly related to savings daylight time. Even with tightly controlled error processing, uncertainty in head data for some specific events clearly exceeded anticipated errors levels. These anomalies were attributed to field sampling and or field recordation errors. Time-offset errors should not impact slope-elevation or flow models that were based on synchronized input data sets. Indicated in Figure D1, periods of malfunction were deleted from the record and occurred more frequently at LCS9.

The analysis in Table D1 using weekly data represents an independent verification of water level meter accuracy for the period of record and was collected using an electronic meter by two different technicians. Overall the gage at CGS1, located in a 1.5m deep pool, was more stable. The data excludes six weekly events where large systematic errors were identified at both sites the same day, likely attributed to the wrong sampling date or clock offsets. These type errors would not be expected to affect inter-gage comparisons where these events were included in flow regime model development in Figure D2. Along with omitted periods of malfunction, the weekly data also excluded three events at CGS1 and two events at CGS2 where errors were high and the respective meter was operating at or above the limits of rated compressor capacity. Compressors for the bubble line meters used were only rated to 3m of true head but typically operated well up to and sometimes above 3.5m to 4m of head. Extreme peaks in Figure D1 are thus qualified by these limitations though this aspect did not inhibit adequate flow characterization for ecological risk assessment purposes. As an example of hydraulic variation encountered for the flood of March 1997, 7.4m of head change occurred at station CGS2/LCS9, covering the 1.5x1.5x1.5m meter house with 1.5m of water.

D.4.3.2 Weekly Stage Data Used in Flow Regime Models

Low, medium, and high flow regime models and development and analysis of stage discharge relations at the 16 gage sites were conducted. In the process, an intensive review of system wide hydrological data and underlying quality was completed. The effort was accompanied by an assessment of uncertainty in head elevation data used to calibrate flow models. The final survey datum control tables presented in Appendix E represent fairly accurate control of measuring point datum for use in watershed based risk assessment. The primary goal of this work was to ensure the capacity to accurately reflect flow conditions associated with various data sets collected under the risk assessment. Flow modeling was conducted to allow for 1) gross characterization of study data sets into low, medium, and high flow conditions (or assignment of flow-invariant status), and 2) to provide estimates of instantaneous flow during routine monthly water quality sampling to allow for determination of chemical loading. Confidence in benchmark elevations was maintained by onsite control errors of ±0.003 mm to 0.006 mm and between site errors of ±0.03 m to 0.3 m, covering estimates of stream depths and slopes. In the latter case, further calibration resulted in between site errors of < ±0.03 m.

Weekly monitoring was conducted from March 1996 till June 1998, and continuous monitoring shown in Figure D1 from May 1996 to November 1997. Errors above 1% were avoided as possible since these can result in relatively large errors in flowrate, particularly at lower flow conditions where stage-discharge relations change rapidly. Assuming accuracy and consistency of continuous station data for given hydrographs, for station CGS1, four weekly event data points were measured at or just above the 1% stage error level. For CGS2, 12 events were found to be between 1% and 2%, 11 events between 2% and 5%, and 1 event above 5%. In comparing these trends to later analysis of CGS1-CGS2 influenced by apparent underreported mine flowrates, note these errors are still centered about averages <1%, indicated in Table D1.

D.4.3.3 Monthly Stage Data Used in Flowrate Models

Water quality was sampled approximately monthly for a suite of metals and other parameters between April 1996 and October 1996. Sampling was limited to collection of water quality indicators during the summer of 1997. For monthly water quality sampling, head measurement error was generally doubled to ± 6mm to 12 mm. This was attributed to multitasking associated with these events, increased speed at which measurements were collected by water quality sampling technicians using metal tapes, and the reduced experience of those technicians in collecting head measurements. Weekly head measurements were all collected with an electronic meter representing the only task performed at each site during those visits. As an example of errors possible, assuming 6 m of tape is lowered from the bridge, an induced error in angle of 2.5 to 5 degrees between the tape and true gravity-induced vertical will result in measurement error of 5.7 mm to 23 mm, respectively. This example emphasizes the care that needs to be taken in head data collection. Head measurement error increases with increased air-space between the bridgedeck and free surface when small errors in head also typically result in relatively larger errors in estimated flow, compounding each aspect.

Comparing weekly same day head data where available to water quality event sampling data, the water quality sampling technicians exhibited a relatively consistent 0.03m stage measurement error (σ = 0.02m; n = 45) resulting in under predicted flows. Site specific errors appeared to depend on depth to water where for the three sites with the smallest air space, TS3A, TS11, and TS16, errors were less, typically around 0.015m. These conclusions were based on sampling events for June, July, and October 1996, and January, February, March, April, and June 1997, excluding August 22, 1996 due to highly variable flows. Similar patterns were observed with average error of 0.03m to 0.04m comparing CGS1 and CGS2 instantaneous data to water quality sampling event data for those two stations when time of day was available. A standardized adjustment of 0.03m was therefore applied to directly measured stage for all water quality events where the adjustment was halved for TS3A, TS11, and TS16. All water quality head data was sampled via metal tape and weekly data was sampled using the electronic static water level meter. These results showed through direct experience that careful measurement for both techniques is necessary to obtain consistent stage error levels below 10 mm. One technician involved in approximately half of the sampling events appeared to show slightly higher overall levels of errors. Relative differences were small and no attempts were made to distinguish between samplers.

At station CGS1/LCS5B, for water quality sampling events with manual gage readings, associated flowrates were underestimated on average 18% (σ = 7.9%; n = 15) compared to concurrently rated CGS1 flow. Recording actual time of day was missed for water quality sampling in the first half of 1997. For these dates, noontime values at CGS1 were used where referenced to concurrent CGS1 instantaneous data. Without time estimated monthly water quality sampling stages, the CGS1 average flowrate recovery error was 16% (σ = 9.9%; n = 9) before correction. In comparison, after adjusting for technician sampling error, flowrates at CGS1/LCS5B were on average 5.7% (σ = 6.2%; n = 15) for the full set and 6.2% (σ = 7.0%; n = 9) excluding sampling event data with time of day estimated. These values give an independent assessment of flowrate accuracy at gaged sites for water quality sampling events studied here and were based on the use of the same stage-discharge rating curves.

D.4.4 NPDES Flowrate Data

To also understand uncertainty associated with independent daily flow data reported by SOCCO for active underground Mine Nos. 2 and 31 that reside within Leading Creek, a review of those operations was conducted. The mines are shown in Figure 3.1 where Mine No. 2 discharges to Ogden Run upstream of monitoring station TS6, and Mine No. 31 discharges to Parker Run upstream of station TS11. Daily Mine No. 31 flowrate reported by SOCCO is compared to weekly data collected at station TS11 in Figure D3. Active mine data discussed here including flowrate, rainfall, and water quality was accessed through monthly data found in EPA's Mor database as provided by SOCCO personnel. Permit required flowrate data provided by SOCCO to EPA is only reported typically on non-holiday weekdays.

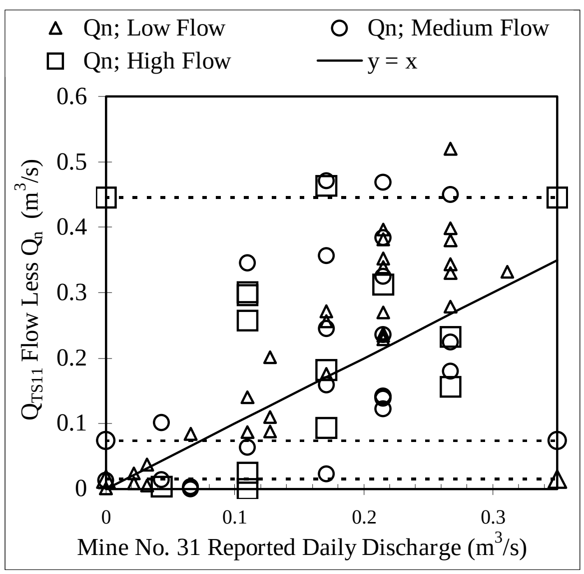


Figure D3 – Active Mine No. 31 Reported Daily Flowrate Compared to Expected Flowrate

Figure D3 indicates active Mine 31 treated process wastewater effluent flows were often underreported during the study. To avoid potential bias associated with mine No. 2 and No. 31 effects on station CGS1, natural discharge Q_n at TS11 was estimated from the low-medium flow condition model ratio and rated flow at station LCS2 collected the same day. Drainage area of TS11 is 14.8 km^2 and that of LCS2 is 34.5 km^2, making them more compatible with respect to any potential heterogeneity not accounted for in this analysis. Dashed lines indicate natural average low, medium, and high flow conditions for TS11 where individual data points were marked for comparison. This analysis excluded estimated weekend data, all dam affected data, and events with daily rainfall total > 0.5 cm, 48 hr total rainfall at > 1 cm or 72 hr total rainfall > 3.8 cm. Derivations for Q_n at TS11 using CGS1 data and CGS1 data adjusted for Mine No. 2 flow gave similar dispersion about y = x except for one high flow event where reported flow was negligible compared to rated Q_{TS11} and the estimated Q_n portion. Average recovery on estimates of $(Q_{TS11}-Q_n) > 0$ between approaches using adjusted CGS1 data and LCS2 were 93.4% (σ = 22%; n = 28) and 104% (σ = 32%; n = 18) for low and medium flow conditions, respectively. An additional check on potential natural variation in discharge rates across the watershed used a rain data filter with threshold values halved which resulted in similar departure from unity for the three comparative examples studied.

It would be expected that measured flows instream would be less than mass balance would predict due to evaporation, losses to groundwater due to increased head pressure *in situ*, and overall reduction in natural baseflow as a result of decreased floodplain groundwater table slopes. Analysis here points out that use of NPDES data can be severely limited in watershed studies as typically applied in mass balance analysis of watershed systems such as EPA's BASINS model application currently used to assess Total Maximum Daily Loads (TMDLs). Because TS11 flow rates were greater than predicted based on drainage area models for low, medium, and sometimes even high flow conditions, mine flows appeared to be inaccurately reported on a consistent basis. The flows directly measured at TS11 could not be attributed to natural drainage or other NPDES sources. Accuracy of TS11 rated data was supported by flow condition models representing natural discharge conditions where corrections for the lower mainstem were also made for active mine inputs, previously detailed in Appendix C.

Watershed hydrological assessment was also greatly complicated in Leading Creek by mainstem flow conditions subject to highly variable, unreported weekend discharges from Mine No 31 and sometimes Mine No.2. A pattern of increased effluent discharge rates between reported rates for Friday and Monday flows were often observed where NPDES permits required only weekday recordation. The pattern was observable through station CGS1/LCS5B stage data that is related to Parker Run and active mining discharges via a slope-stage-discharge relation (Linsley et al, 1978). Induced increases in stage on the mainstem of 0.03m to 0.15m can occur. These factors together rendered mass balance approaches using active mine flowrate as an independent variable untenable. This was one of the driving forces behind the subsequent flow condition assessment approach utilized instead, along with non-mass balance approaches for extrapolating extreme low flow rating information discussed in Appendix C.

To further evaluate the relationship between Mine No 31 flowrates a comparison was also constructed between gage CGS1 and corrected CGS2 results adjusted for Mine No. 31 reported daily discharges. Based on independent assessments of flow regime models, these data should have also reflected consistent flow-area relationships, particularly if rainfall heterogeneity was accounted for in the analysis. The daily Mine No. 31 flowrates were again taken from data reported to EPA's Mor database. Estimated weekend data assumed as Friday's level was evaluated separately. Figure D4 excludes days with daily rainfall total > 0.5 cm, 48 hr total rainfall at > 1 cm or 72 hr total rainfall > 3.8 cm. Analysis also excludes periods of malfunction, backwater days plus the day before and day after, and days where CGS1 or CGS2 daily average water temperature was < 1^{o}C or minimum temperature was < 0.5 ^{0}C. Results similarly indicated

Mine No. 31 flows were underreported. Mine discharge rates were observable instream for low and medium flow conditions or Q_{CGS1} < 1.5 to 2 m^3/s, representing a 5:1 ratio of stream flow to mine flow. Analysis excludes drought data in fall 1997 where flow values at LCS9 fell below 0.1 to 0.2 m^3/s and active mine flows were low with highly variable daily patterns. The latter period was too difficult to accurately assess slope-stage-discharge relations induced at CSG1.

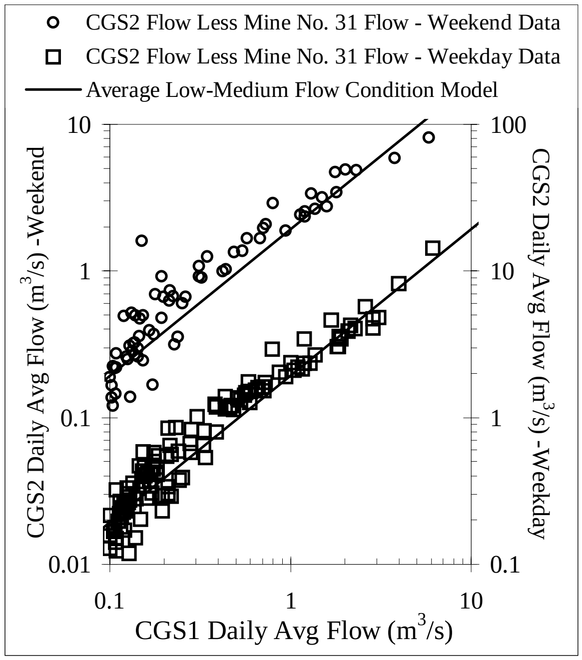


Figure D4 – Evaluation of Mine No. 31 Flowrates via Comparison of CGS1 and CGS2

This exercise of halving the rainfall filter for analysis presented in Figure D3 indicated that the rain filters used in Figure D3 and Figure D4 were conservative and successfully identified events where stable recessional runoff patterns or baseflow discharge prevailed. Combined analysis of Figures D3 and D4 also indicated the degree to which Mine No. 31 flows can dominate the mainstem during lower flows and drought, and partially during medium flows, but with less significant affect during high flow conditions. This analysis indicated errors in reported daily average flow from Mine No. 31 of up to 200%, where over reporting errors of 50 to 100% could also be observed but were less frequent. Shown in Figure D4, the effect of increased flowrates observed on weekends via the continuous stage record at CGS1 were notably modest in comparison to overall noise observed in actual daily average data. This is seen in the relatively similar patterns expressed in both weekday and weekend curves evaluated. This is difficult to directly assess due to short time frames for weekend excursions, and interspersed data gaps induced by rain filters applied. Curves in Figure D4 are not matched data sets. Averages over each flow condition range though do impart some reliability in this conclusion

D.4.5 Hydrologic Conditions for Water Quality Sampling Events

Prior to modeling flowrates at ungaged sites for specific water quality sampling events, an evaluation of hydraulic and climate data was conducted. This allowed for proper evaluation of potential heterogeneity in rainfall patterns and to understand its relationship and expression instream at various stations across the watershed. The methodology undertaken evaluated water quality stage data sets collected at times of sampling and compared these to independent weekly data also being collected, which for the most part occurred on the same day or an adjacent days. Monthly water quality sampling was conducted in almost all cases across two days due to the large number of sites monitored which included 28 stations in the watershed and periodic sampling of reference watersheds described in Chapter 3. Reference watersheds though were not directly characterized in the hydraulic analysis. Except for added sampling events for water quality indicators, SOCCO and American Electric Power scientists and technicians conducted all water quality sampling. Other SOCCO engineering staff collected weekly stage data, and there was no attempt placed on synchronizing these activities.

Hydrologic conditions for water quality sampling events are presented in Table D2. Flow regime encountered for these events is presented for both cases of assignment based on flowrate at CGS1/LCS5B, and assignment by site-specific flowrates encountered where the latter excludes consideration of TS11. Q_n implies natural flow was estimated from event flow condition models and concurrent data at station CGS1 where CGS1 was first adjusted for Mine No.2 discharges. Daily rainfall totals are given for four gages along with daily totals added to total rainfall for the previous 48 hours. From these, daily rainfall can be determined for each of four days leading up to and including the two-day sampling events. Flow condition was assessed throughout the watershed for various data collection activities. For Leading Creek, medium average monthly flows can be experienced typically in spring (April to June) and high flows during winter (December to March). Daily high, medium, and low flow conditions can occur in any month. For this region of the United States, the high flow condition will typically include bankfull flows for study periods > 1.5 to 2 years (Leopold et al, 1964). Discussed in detail in Appendix C, this study of Leading Creek encountered unusually high rates of flooding with 9 bankfull exceedences encountered at LCS5B. Together with periods of drought, overall average short-term data mimicked long-term trends.

Similar for weekly averaged data events outside of CGS records, representative daily flow condition at LCS5B was estimated in April 1996 from a single head value collected the same day. Indicated by asterisk in Table D2, the 11/96 event exhibited borderline low flow for stations LCS1 and TS14, and for 4/97 at LCS1 alone. For the same timeframe weekly stage sampling indicated lower-end medium flow conditions at other stations. LCS1 medium flow was attributed to quarry discharges for the pulse storm event monitored in 10/97. Storms 7/24/96, 1/22/97, and 6/26/97 occurred after sampling was completed and rainfall on 10/23/96 fell over a 3-hour period before dawn. Events with days split by over night storms included 8/96, 12/96 and 3/97. The April 1996 event was also characterized similarly but continuous 5-minute data was unavailable for detailed evaluation. Other monitoring events in Table D2 were characterized by relatively stable runoff or baseflow patterns in widespread recession. Modest backwater effects were observed during sampling on the first day at stations LCS9 and LCS10 for October and December 1996 events.

D.4.6 Flowrate Modeling for 2-Day Water Quality Sampling Events

Instantaneous stage collected at gaged sites during water quality sampling events in Table D2 were rated for instantaneous flow rates and segregated on a daily basis, creating associated "event-day" groups of flowrate data. Like statistical based models shown in Figure D2, initial regime-type flow-area models were created and their stability assessed along with climate data to characterize potential run-off heterogeneity. Two-day events with stable conditions emulating either near-linear or consistent non-linear behavior were combined for both days. From these data sets, event-day models were generated which allowed estimation of flow at ungaged sites. Less stable events with uncombined event-days were then further evaluated on a daily basis to assess individual event-day behavior. Prior to initial and final model development for use at ungaged sites, a thorough evaluation of input data was conducted. Paralleling Table D2, final event-day models are given in Table D3 where $Q_{natural} = Q_n = aA_d^b$ and A_d = drainage area.

Table D2 - Hydrologic Conditions for Water Quality Sampling Events – Low (L), medium (M), and high-(H) flow conditions for gage stations based on a) spatially-invariant assignment by gage CGS1 and b) instantaneous site-specific rated flows. Asterisks indicate data is qualified.

Watershed Flow Condition (m^3/s)						Total Rainfall (cm) at CGS Stations and Active Mine Sites							
Water Quality Sampling Event Date	CGS1 Daily Avg Flowrate	Active Underground Mine Daily Avg Discharges			Range of Site Specific Flows	Continuous Gage Stations				Active Mine Manual Gages			
						CGS1		CGS2		Mine No. 31		Mine No. 2	
		No. 31	TS11-Q_n	No. 2		Daily	72Hr	Daily	72Hr	Daily	72Hr	Daily	72Hr
4/29/96	M ; (1.5)*	0.11	0.23	0.063	M, H	-	-	-	-	0.61	0.61	0.64	0.89
4/30/96	H; (15)*	0.11	-	0.063		-	-	-	-	2.54	3.15	1.32	2.21
5/29/96	H ; (20)	0.04	No.31	0.025	M, H	0.36	3.68	0.51	2.24	0.71	4.55	0.64	4.29
5/30/96	H ; (6.1)	0.04	<< Q_n	0.006		0	1.24	0	0.81	0	3.18	0	2.39
6/26/96	L ; (0.25)	0.21	0.17	0.13*	L	0	1.27	0	0.10	0	2.08	0.13	1.60
6/27/96	L ; (0.18)	0.21	-	0.006		0	0	0	0.08	0	2.08	0	1.60
7/23/96	L ; (0.16)	0.17	0.26	0.002	L	0	1.02	0	0	0	1.24	0	2.95
7/24/96	L ; (0.13)	0.17	-	0.002		1.04	1.42	0	0	0	1.24	3.12	3.99
8/21/96	L ; (0.18)	0.27	0.34	0.002	L, M, H	1.78	1.78	-	-	0.18	0.18	1.07	1.07
8/22/96	H ; (4.8)	0.27	-	0.001		0	1.78	-	-	1.55	1.73	0.94	2.01
9/18/96	M ; (1.7)	0.27	0.31	0.010	L, M, H	0	5.74	-	-	0	6.15	0	3.86
9/19/96	M ; (0.65)	0.27	-	0.006		0	0.69	-	-	0	3.66	0.05	0.10
10/23/96	L ; (0.26)	0.21	0.14*	0.003	L	0.38	0.38	-	-	0.43	0.46	0	0.38
10/24/96	L ; (0.26)	0.21	-	0.001		0	0.38	-	-	0	0.46	0.28	0.64
11/19/96	M ; (1.6)	0.21	0.22	0.004	L*, M	0	0.97	-	-	0.10	1.17	0	1.02
11/20/96	M ; (0.98)	0.21	-	0.003		0	0.36	-	-	0	1.17	0	0.20
12/16/96	H ; (2.1)	0.21	0.32	0.002	M, H	0.84	0.84	-	-	0.03	0.03	0.69	0.69
12/17/96	H ; (15)	0.21	-	0.019		0.97	1.80	-	-	2.01	2.03	1.65	2.34
1/21/97	M ; (0.41)	0.17	0.32	0.003	L, M	0	0	-	-	0	0	0	0
1/22/97	M ; (0.61)	0.17	-	0.004		0.33	0.33	-	-	0.15	0.15	0.25	0.25
2/18/97	H ; (1.9)	0.21	0.47	0.002	M, H	0	0.03	-	-	0	0.05	0	0
2/19/97	H ; (1.9)	0.21	-	0.002		0.05	0.05	-	-	0	0	1.52	1.52
3/25/97	M ; (1.4)	0.21	0.16	0.005	M, H	2.03	2.03	-	-	0	0	2.44	2.44
3/26/97	H ; (17)	0.21	-	0.006		0.08	2.11	-	-	0	0	0.13	2.57
4/15/97	M ; (0.72)	0.21	0.20	0.11	L*, M	0	0.05	0	0.08	0.08	0.08	0	0
4/16/97	M ; (0.63)	0.21	-	0.057		0.41	0.41	0.43	0.43	0	0.08	0.41	0.41
5/14/97	M ; (0.71)	0.24	0.22	0.001	L, M	0	0.03	0.13	0.13	0	0.10	0	0
5/15/97	M ; (0.59)	0.27	-	0.002		0.13	0.13	0.10	0.23	0	0.08	0.08	0.08
6/25/97	L ; (0.15)	0.27	0.40	0.001	L	0.00	0.00	0.00	0.15	0	0	0	0
6/26/97	L ; (0.16)	0.27	-	0.013		1.09	1.09	2.29	2.29	1.60	1.60	1.55	1.55
10/24/97	L ; (0.11)	0.11	0.16*	0.001	L, M*	1.27	1.27	1.63	1.63	0	0	0.79	0.79

The event-day models in Table D3 were based on gaged sites and primarily intended for use in assessing flowrate in subsheds between 1.2 km^2 and 19 km^2, representing ungaged tributaries monitored for water quality. Poor model fit was associated with a small data set characterizing heavy overnight rains that split the March 1997 event. On the second day, upper mainstem stations and ungaged tributaries were sampled in the morning where LCS1 and stations 8 to 20 km^2 were approaching or just passing crest. Errors of up to ½ order of magnitude may be expected for that event. The curve shown represented a balance between the largest size subsheds of 19 km^2 to be estimated and smaller stations already on the falling limb of the storm hydrograph. The split low flow event-day model given for 8/21/96 utilized TS16 data from the week of 8/14/96. No rains fell in the interim and flowrates closely agreed with 8/21/96 data in other smaller sized subsheds that were sampled. An outlier, TS14 was omitted from the 11/19/96 model due to 160% departure from trend which was supported by similar departure measured at the site two days later. This was attributed to lower rainfall in the subshed. With the data retained, the model significantly under predicted flows in smaller subsheds measured one day later, where further recession had occurred.

Table D3 – Leading Creek Water Quality Sampling Event-Day Flow Models

	Q_n (m^3/s) = a*[Drainage Area (km^2)]^b			
Date	**a**	**b**	**R^2**	**n**
4/29/96	2.17E-02	0.779	0.921	8
4/30/96	1.08E-01	0.971	0.899	4
5/29/96	1.13E-02	1.320	0.954	5
5/30/96	1.15E-02	1.210	0.994	9
6/26/96	1.15E-03	1.050	0.935	9
6/27/96	1.15E-03	1.050	0.935	9
7/23/96	1.04E-03	0.988	0.963	12
7/24/96	1.04E-03	0.988	0.963	12
8/21/96	5.70E-04	1.060	0.924	6
8/22/96	1.13E-03	1.410	0.879	7
9/18/96	5.25E-03	1.090	0.953	7
9/19/96	4.57E-03	0.961	0.990	5
10/23/96	1.05E-03	1.080	0.973	12
10/24/96	1.05E-03	1.080	0.973	12
11/19/96	2.09E-03	1.260	0.879	8
11/20/96	4.57E-03	0.961	0.990	5
12/16/96	2.79E-03	1.270	0.894	6
12/17/96	5.56E-02	1.270	0.974	5
1/21/97	8.85E-03	0.835	0.900	14
1/22/97	8.85E-03	0.835	0.900	14
2/18/97	7.37E-03	1.060	0.974	14
2/19/97	7.37E-03	1.060	0.974	14
3/25/97	1.64E-02	0.893	0.942	9
3/26/97	8.02E-02	0.439	0.359	5
4/15/97	6.15E-03	0.939	0.971	14
4/16/97	6.15E-03	0.939	0.971	14
5/14/97	3.35E-03	1.080	0.984	14
5/15/97	3.35E-03	1.080	0.984	14
6/25/97	3.66E-04	1.190	0.846	12
6/26/97	3.66E-04	1.190	0.846	12
10/24/97	6.76E-04	1.050	0.944	7

Seen in the event-day models, some days near-linearity prevailed (b = 1), other days smaller subsheds exhibited higher unit area run-off (b < 1) ahead of the mainstem hydrograph, and other days smaller subsheds were found in recession while the mainstem was still cresting (b > 1). This is not analogous to hydraulic geometry the way bankfull is to mean annual flow, but instead represents a fixed geometry in space and its temporal distribution describing non-flood flowrate dynamics.

D.4.6.1 Data Used in Developing Event-Day Flowrate Models

For water quality sampling events, outlier stage values were identified using comparisons with CGS and weekly data collected concurrently during relatively stable flow conditions on the same day or on adjacent days. In seven cases, large stage errors observed between 0.15m to 1m were attributed to unit increment reading errors (e.g. 1ft.) or reordered digits in recordation. In two cases small errors of 0.1m observed were attributed to suspected surrogate measuring point confusion and poor measurement. The latter case was for site LCS5B where the more reliable weekly reading was used to avoid error in TS11 natural drainage calculations based on LCS5B. A final correction was made for TS16 using March 25, 1997 same day data where flowrate for water quality sampling was 230% above expected rates at this site when otherwise stable recession patterns were observed across the watershed. August 1996 TS11 stage was missed and CGS2 rain data was lost due to flooding in March 1997. Earlier CGS2 rain data was lost to Meigs County roadside lawn mowing activities tied to intermittent meter behavior and damaged buried wiring eventually discovered and replaced after the flood of 1997.

Detailed in Appendix C, stage-discharge ratings at LCS3 and TS11 were impacted by transient beaver activity. The rating curves were developed for non-impacted conditions. The rating for TS3A represents constant dam influence. To ensure station rating curves and flow models were not correspondingly biased, the periods of dam influence at LCS3 and TS11 were omitted from flow regime model development in Figure D2, as was a period representing transient dam removal at TS3A observed in fall 1997. Low flow predictions for water quality sampling events at these sites were evaluated for potential dam influences at each site, for each event and time period as appropriate. While some added error was likely introduced due to minor slope effects, this method allowed use of instantaneous head data collected during water quality sampling at LCS3, best representing actual site conditions encountered at that site. Data for event-days at TS11 could not be handled in this manner. Rated stages were discarded, and treated on a case-by-case basis. Monthly water quality data on Five Mile Run was collected at station TS3B, below another larger subtributary downstream of the actual gage station at TS3A. A unit drainage area ratio of 1.56 was used for estimating all flows at TS3B from flow measured at TS3A.

For all water quality sampling, 233 instantaneous measurements of stage at 15 gaged sites were available for 16 sampling events evaluated. Of these, 23 missing data points were estimated from same day or adjacent day weekly data collected concurrently under stable watershed flow conditions. TS11 stage during October 1996 and 1997 low flow events was influenced by beaver dam activities. Flowrate for these events and comparative estimates in Table D2 were based on increases observed between stations LCS5B and LCS6, accounting for drainage area above and below station TS11. Dam affected data for station LCS3 in October 1996 was based on direct flowrate measurements during that week. Seven missing data points for gaged sites were modeled using the event specific flow models, in addition to two points at LCS9 modeled similarly due to backwater impacts on measured data. Missing data at gage sites was in most cases attributed to time delays in reestablishing reference points disturbed by vandalism and in some cases the technician simply missed collection of stage data.

D.4.6.2 Low Flow, Backwater, and Adjustments for Mine Flows

Shown in Table D3, event specific, day specific flow models were developed for estimation of instantaneous flowrates at ungaged sites and gaged sites with missing data. Increased variability of data at low flow was observed in averaged flow condition models shown in Figure D2, as well as in the underlying event specific weekly models. Variability, especially in tributaries, was often pronounced at levels below baseflow measured in September 1996, particularly for AML tributaries. For these reasons, event specific low flow models for June, July, and October 1996 and June 1997 events were based on combined data over two days which were characterized by stable flow conditions across the watershed. For these models, low flow models excluded TS3A and LCS1 data biased by quarry operations and muskrat dam activities, respectively. For similar benefits of increased data input, 2-day models were also used for stable runoff conditions associated with events in January, February, April, and May 1997.

For the October 1997 event, TS8A on Mud Fork was dry, TS8B flowrate was below practical quantification, and sampling was extended up to midnight to visit all stations for the pulse rainfall event after drought. Observed at extreme low flows for weekly stage monitoring was the drop-off of anticipated flowrate in the lower mainstem below LCS6. Often noted was the apparent sinking of added Mine No.31 flows during drier periods. For the 10/97 event the lower mainstem was avoided in modeling smaller subsheds where the situation was equally complicated by dam influence at TS11. The flow model for 10/97 therefore excluded LCS1, TS3A, TS8B=0, TS11, and LCS6 to LCS10. The model retaining data at LCS7, LCS8, and LCS9 was $y = 7.7x10^{-4} (A_d)^{1.0}$ ($R^2 = 0.96$; n=10).

For final flowrate calculations used for input into curve fitting shown in Table D3, station TS6 was adjusted for addition of Mine No.2 reported flowrate after $Q_{natural}$ was determined, except for the June 1996 event. For this event specific conductivity was 0.52mmhos/cm and the following month it was 0.55 mmhos/cm, further indicating there was no local effect of high discharge rates reported from active Mine No 2 for that event. This was supported by normal conditions of high specific conductivity when Mine No. 2 discharges significant flows, as shown in Figure 5.2b in Chapter 5. Reported Mine No. 2 and estimated Mine No 31 flow inputs shown in Table D2 were added to the only event model estimate used for Q_n on the lower mainstem for missing data at LCS6 on 9/18/96. All event models in Table D3 excluded data points for TS11.

For water quality sampling, measured stage data at station LCS10 was also not used in models or load estimates. LCS10 flow data was based instead on mass balance addition of flowrates at LCS9 and TS17 adding a unit drainage area correction of 1% to account for ungaged drainage area downstream. LCS10 had the largest headspace between the bridge reference point and water surface, on average 3m more than LCS8 and LCS9. LCS10 flow data was based on upstream data to avoid uncertainty associated with stage measurement during water quality sampling. Station LCS10's rating curve was poor and the site was often subject to influences of variable backwater from the Ohio River during low and medium flow conditions. Estimated data for 12/16/96 closely agreed with the weekly event model and rainfall patterns for the previous week representing similar stable recessional flow several days after rainfall. Similar slope in the event-day model was again coincidentally observed the next day for the 12/17/96.

D.4.6.3 Active Mine Discharge Rates for Water Quality Sampling Events

Flow condition models presented in Figure D2 and Table D3 evaluated natural discharge conditions. The models incorporated an offset adjustment for reported Mine No.2 daily flow data. The Mine No 2 discharge represents an overflow AMD treatment mechanism usually associated with rain and increased infiltration following runoff events, particularly in spring as seen in Figure

D1. Exhibiting a modest time lag from peak rainfall on the order of hours to days, the impacts to the receiving mainstem are often negligible due to relatively higher natural runoff discharge rates. Increased influence on certain days may occur, as evidenced in Table D3 for sampling events in June 1996 and more modestly in April 1997. Accuracy of reported discharge at Mine No. 2 is not well characterized. For the June 1996 event, the No. 2 reported flowrate represented a single high daily value among otherwise low discharge rates before and after the event. The presumed flow input could not be observed though at CGS1 until after sampling on June 28th. With adjustments for mine No.2, the event model determined was Q_n= 0.0016A_d^{91} (R^2 = 0.94). The measured CGS1 value adjusted for Mine No.2 was 50% of the modeled result. These aspects resulted in significant departures from the expected average low flow condition trends characterized in Table D2. Due to the disparate event model parameter estimates for the June 1996 event, no adjustments for reported Mine No. 2 discharges were made. Born out by final analysis, Mine No.2 would not be expected to play a significant role in water quality for the periods sampled. Occasionally a day lag in Mine No. 2 rainfall data was evident, perhaps explaining some of these discrepancies.

Mine flows were evaluated for concurrent weekly stage measurements represented in Figure D2. Between March, 1996 and June 1998, in units of m^3/s average Mine No.2 discharge rate during low flow was 0.012 (σ = 0.026; n = 50), for medium flow it was 0.015 (σ = 0.028; n = 33), and for high flow 0.053 (σ = 0.049; n = 39) was reported, respectively. For Mine No. 31, discharge in m^3/s for low flow conditions was 0.162 (σ = 0.086; n = 50), for medium flow 0.165 (σ = 0.079; n = 33), and for high flow 0.150 (σ = 0.090; n = 39), respectively. SOCCO NPDES discharges discussed here consider only process wastewaters associated with outfall 001 at Mine No. 31 and outfall 005 at Mine No. 2. The analysis excluded other managed stormwater outfalls, all of which employ a 10-year recurrence interval design basis. The analysis also excluded negligible treated sanitary wastewater discharges from active mines, and the STP in Rutland, Ohio. At Mine No.31, based on Landsat5 analysis of identified gob and spoil areas, managed footprints comprise <6.8% of the Parker Run subshed above station TS11, and the make-up water reservoir comprises <2.7%. Even with managed stormwater, the areas would detract only modestly in differences between reported discharge rates and measured rates adjusted for Q_n shown in Figures D3 and D4.

For water quality sampling events, measured TS11 flowrates were also compared to modeled results for natural discharge based on CGS1 data and uncorrected water quality event stage data. Using Q_n based on a simple assumption of unit drainage area ratio with CGS1 data, the average error estimate for Mine No. 31 discharge rates was 15% (σ = 51%; n = 13) over actual reported values. This assumes omission of beaver dam influenced data, the May 1996 event when natural flows dominated Parker Run and no discharge at Mine No. 2 for the June 1996 event. Excluding the November 1996 event with an error of 131%, average error in reported Mine No. 31 discharge rates was 5.0% (σ = 38%; n = 12). Incorporating corrections for monthly stage data collected during water quality sampling events, over-reporting error estimates were -10% (σ = 37%; n = 15) with October 1996 and 1997 data and -14% (σ = 33%; n = 13) without. Shown in Table D2, using similar data but with Q_n estimated from flow event models in Table D3, the average over-reporting error estimates were –12% (σ = 34%; n = 15) with October 1996 and 1997 data -16% (σ = 30%; n = 13) without. In summary, fair agreement was found between reported flow rates at No 31 and those measured at TS11 during some water quality sampling events, but with some event errors ranging upwards of ± 50% and one event with error up to 131%.

D.4.6.4 Lower Mainstem Stability During Water Quality Sampling

Figure D5 depicts relative stability, consistency, and an overall average run-off heterogeneity or departure from normal flow regimes across the entire water quality monitoring program for the lower mainstem. It is represented by matched daily average flowrates at stations CGS1/LCS5B and CGS2/LCS9. This data was corrected for Mine 31 inputs, and was also subject to unfiltered rainfall events delineated in Table D2 where the two backwater events were excluded. The two points of departure in Figure D5 are also related to the only event in which LCS5B was sampled on 8/21/96 and split with LCS9 sampled the next day, in this case after heavy rains. For reference and comparison, a similar average low-medium flow regime ratio is shown with b = 1.045. This curve presents data for both days of each event, excluding March 1997 when CGS2 was offline. The second day of the split event was controlled by LCS5B cresting upstream at 7 a.m. and LCS9 at 5 p.m., with net average relative weight given to LCS5B for the day. For the only severe spilt event noted, the first day saw a tendency for drop-off in flows at LCS9 attributable in part to either sedimentation or, less likely, over compensation due to over-reporting at Mine No. 31.

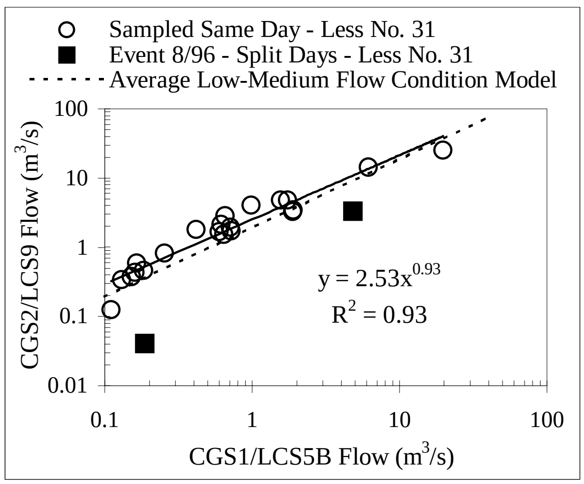


Figure D5 – CGS1 and CGS2 Daily Average Flowrates for Water Quality Sampling Events

Available data sets make distinctions difficult to assign on any given day, except through averaging over time. A drop-off for drought in July 1996 was not seen perhaps due to underreporting, but was seen in the drought in October 1997. The phenomena of drought exacerbated by ASML sedimentation on the lower mainstem was only readily established during drought with co-occurrence of mine shut-off, as discussed in Appendix C. Effectively increased frequency in drought due to massive AML sedimentation though was supported by similar behavior on most AML tributaries unaffected by active mining.

D.5 DISCUSSION

D.5.1 Flow Regime and Instantaneous Flowrate Modeling

Hydraulic descriptions of the Leading Creek watershed were defined in Appendix C for flow regimes spanning low flow, medium flow, and high flow conditions. These were intended for use in separating water quality concentration data based on flowrate at a central gage, or based on site-specific flowrates at several gages. A single flow regime for example might be identified and used at ungaged or gaged sites based on known flowrate at any one station. This could be done to assign the associated regime attribute to all sites collected during a single event. If an assumption that unit-area run-off did not prevail for the given time frame, error may be introduced. The problems faced in estimating instantaneous flowrates at ungaged sites are analogous to this error. In comparison to average models based on sampling 30 to 50 events, for a single event far less success will be realized in reaching across flow regimes when heterogeneity in unit area run-off prevails.

For the centralized case, knowledge of heterogeneity for a single event needs to be quantified to higher degree where it was seen in Appendix C that large sample populations sufficiently dampened variation in unit-area trends over time. Conceptually this can be further extended by considering several group assignments defined by several gages and an A_d ratio criteria. A_d for ungaged sites would then offer a capacity for initial pre-selection, and subsequent regime assignment. Modest or large variation observed at a gaged site for a given event may strongly bias final results. Equally problematic, sampling may be missed or split for two related sites across two days. It would be desirable then to minimize the number of models considered within a practical time-step. In any framework it can be seen that using spatial location (e.g. the closest gage) without consideration of temporal distribution likely leads to greater errors for most sampling events then event-day models. Here drainage area was used to advantageously assimilate time-variant data. These concepts form the basis for development of event-day models presented.

Even if gross segregation were successful in discrete classification of low, medium, and high flows under non-linear conditions, this still only addresses concentration data sets. A critical task in applying the regime approach to ecological risk assessment data sets would remain in estimating accurate flowrates for a given timeframe at ungaged sites, for later separation into low, medium, and high regimes. Loading rate statistics of those populations otherwise could only be assigned on discrete levels, by average flowrate for example, with no knowledge of inter-regime variation. This problem presents itself if analysis and segregation of unit area loading rates are of interest in addition to examining patterns in mass concentration.

Shown in linear modeling results in Appendix C, for average conditions, a single gage could be used with almost equal success in predicting flowrate across all other gaged and ungaged sites. In context of the rational method, to separate average flowrate conditions from average concentrations for each regime would need similarly large sample populations of weekly water quality data across all subsheds which is not feasible. In this case, spatial variation in land use would also have to be addressed in the gaged network. Similarly, by default the gaged network is well defined for handling conditions of rainfall and associated heterogeneity in run-off rates for less ambitious monthly water quality sampling programs. The problem enters then when hydrographs across ungaged sites are subject to non-linearity with respect to drainage area, or phrased another way, there is heterogeneity in unit-area run-off for monthly sampling. Quantifying this problem in Leading Creek was accomplished on two levels. On the simplest level for mass concentration, it was desirable to retain a capacity to segregate flow-variant water quality monitoring data into groupings reflective of hydraulic dilution concepts. Concepts of low, medium and high flow were devised to distinguish between hydrologic processes associated with baseflow (or groundwater) discharge and overland run-off. On a second more detailed level, models were developed here to predict instantaneous flowrates at all ungaged sites for monthly water quality sampling events conducted over a two-year period.

D.5.1.1 Error and Hydrograph Stability in the 2-Day Event Sampling Program

Flowrate data was first classified with respect to uncertainty in stage data used to generate flow ratings. Average errors encountered in instantaneous head measurement data sets for continuous gage stations after calibration were ± 0.02%, shown in Table D1. Subsequent comparison of weekly measurement data set average errors at those stations ranged consistently less than 1% of stage. Errors in stage measurement were controlled in this project to typically ± 0.003 m to 0.006 m for weekly data and doubled to ± 0.006 m to 0.012 m for water quality sampling. These rates were for stage measurements collected from bridgedecks where 689 headspace readings taken at site-specific low flow conditions ranged 1 m to 7 m, excluding LCS10 which ranged up to 9 m. The three stage reading programs (continuous, weekly, and monthly) underwent thorough review to assess anomalies in the data sets. With over two years of monitoring, occasional outliers and systematic errors were observed where quality assurance was critical to success. Identification and evaluation were greatly assisted by independent weekly data collections. The weekly data sets served as a key element in the hydraulic assessment, where continuous monitors were critical to characterization of NPDES sources and short-term temporal distribution of rainfall and flow.

Uncertainty in rated flowrates where based upon confidence in stage-discharge curves developed in Appendix C and shown in Appendix G. Stage-discharge curves for most sites showed high confidence in the spilt lower and upper regime models developed for each site, with qualifications noted. Deviation in raw data was typically < 10% to 20% of flowrate estimates at those water levels. For actual ratings at stations other than CGS1 and CGS2, fewer data points increase uncertainty for gaps not well covered by instream velocity and flow area profiling. For all subsheds, gaged models fit expected parabolic shapes indicating the gaps between field data were likely well described and within similar tolerance. Additional data analysis of water quality sampling events and associated stage and weekly data collected independently allowed reduction of flowrate errors in event-day models. For water quality sampling events with manual gage readings, associated flowrates were underestimated on average by 16% to 18%. Application of an average stage offset due to over-measurement reduced errors between weekly and monthly flowrate data to 6%. Errors in water quality stage sampling were systematic and attributed to use of metal tapes versus electronic meters, speed of measurement, and lack of experience in water quality samplers. These last error rates stated between weekly and monthly data are relative to station rating errors.

For 15 monthly sampling events conducted at all stations for all parameters, 3 events were determined to retain significantly shifted hydrograph positions between most stations sampled. For one of these events described in Figure D5, the lower mainstem exhibited significant departure from flow regime stability. This latter event split across low and high flow conditions, occurred in August, and was attributed to a 1.8 cm rain event, of which 93% fell within 15 minutes. Examining relatively stable events, flow condition assessment at best still represented a gross oversimplification of describing variable pollutant concentration and loads. Notwithstanding, using several samples collected across annual and seasonal cycles, flow conditions adopted here were found to be useful in distinguishing key trends between pollutant concentration, loading, and land use. Smaller subsheds were more transient than larger subsheds and more difficult to characterize.

Seen in Table D3 due to the effects of normal hydrograph procession, event-day specific models of flowrate with respect to drainage area can exhibit steep or shallow curve slopes, depending on crest position residing in smaller or larger subsheds for a given timeframe. Storm patterns initially raise relative flowrates for smaller subsheds and end with elevated rates for the largest subsheds. The power function model eventually returns to the nominal near unit slope during stable recession. The ability of event models to predict relatively accurate flowrates is well supported by hydraulic principles. For runoff patterns not completely seated in stable recession patterns across the watershed, event specific models become superior to an assumption of near-linear drainage area ratio models as presented in Figure D2. At the same time, the event-day models represent a simple, calibrated approach compared to using more complex runoff models like HSPF for assessing instantaneous flowrates in subsheds spanning 1to 400 km^2.

For this application, confidence in instantaneous flowrate estimation is based upon the strength and consistency observed in non-linear and near-linear event-day models. This translates directly to the underlying confidence associated with event-day flowrate model uses in acid mine drainage signature analysis in Chapter 6 and unit-area pollutant loading rates. The models were developed specifically for typical application to ungaged subsheds between 1.2 km^2 and 19 km^2. The two-day sampling program described for Leading Creek could have benefited significantly from collecting head measurements at a few sites or all sites each day of water quality monitoring to ensure better general coverage of drainage area and watershed dynamics. This loss was offset considerably by weekly data collection that usually coincided with water quality sampling activities and would have been avoided had no stage measurements been missed.

D.5.2 Temporal and Spatial Variation in the 2-Day Event Framework

Using a single grab measurement at each site, the disparities in application of this concept to the watershed-scale problem increase the closer actual time of sampling is to peak time of rainfall. Ideally, when attempting to integrate a wide variety of subshed sizes in this manner, one hopes to capture runoff conditions where all sites were sampled on rising limbs, or all hydrographs were in early recession, or all were sampled at time of peak flow. This would be more ideal if these aspects could be captured equally across two-day events, which did not always happen in Leading Creek sampling. With respect to risk assessments based on small data sets, it is not the absolute magnitude of flowrate that holds greatest importance, but consistency of hydrograph position. This assumes a reasonable level of rainfall homogeneity existed over the course of sampling. It is also realized that activity on the landscape is enmeshed to a degree with seasonal variation. A monthly sampling program that uses only single day event sampling is then far more ideal when comparative land use/cover analysis is intended. Understandably, as the number of sampling events decrease, any inter-dependency in hydrograph position and seasonal variation will exert greater influence.

The degree to which concentration is independent of flow rate and unit area flow rate contributes significantly to the power of comparative analysis among subsheds. This is simply a restatement of the rational method (Thomann *et al*, 1987). Flow condition factors are useful for various purposes but may be limited in their ability to assess inter-annual, inter-seasonal, and daily spatial and temporal variation. By virtue of the size and complexity of problems investigated, watershed risk assessment will more often than not be severely restricted in data collection. There can be advantages then to separation of paired time-sequence data by flow condition. Distinguishing between runoff and base flow characterization can assist in identifying interaction of the stream system and the landscape, and its driving mechanisms.

Aside from Figure D2 and Table D3 describing a convention for using direct measurements to establish flow condition and flowrate at any given time *t* in Leading Creek, there are several regional data resources that might also be used for estimation for various sampling events and programs. For instance weather data, or nearby real-time gages operated by the U.S. Geological Survey could assist in this classification process. For meaningful results in land use/cover analysis with several significant sources of stress defined, the size of subsheds studied would appear to need inclination towards smaller subsheds not typically gaged. For the most part this would only limit the utility of mentioned data sources in solving for the instantaneous flowrate problem. Attempting to relate landscape to instream pollution moves holistic investigations closer to the realization that better defined methodologies in network design and execution of sampling programs are needed.

The degree to which flow can be quantified in this way on a given day throughout a watershed depends on the ability to match relative positions of hydrographs site to site. For grab sampling programs, this synchronization or lack thereof will be a direct reflection of the potential bias inherent in assessing runoff conditions with event-day models. Though never realized, synchronized single grab sampling at all sites on similar regions of a normalized unit-area hydrograph would be the ideal pattern desired when attempting to infer relationships between land use and pollutant concentration or load. For randomized grab sampling though, one can only reflect on the stability of each event after the fact and local position. Augmented by one or more nearby continuous stream gages, assessing the actual stability of each sampling event can be improved.

D.5.2.1 Discrete Hydrologic Simulation

For rainfall-runoff analysis, hydrologic simulation time steps can include daily, hourly, 15 minute, and even 5-minute time steps. Simulation in this sense infers a transient component, with transient inputs (Bicknell et al, 1993). A time-simulation model is represented by a continuous time-sequence input data set used to drive the model (e.g. rainfall), which then predicts a continuous time sequence output data set for some set of parameters of interest in the watershed, for example flowrate or stream stage. Both sediment and water flow can be included as output variables of the simulation model, as well as loading of pollutants depending on the actual simulation model chosen. For example HSPF is a 1-Dimensional (1-D) flow routing model, based on uniform flow assumptions, a fair assumption for most alluvial streams. The model though requires a great deal of input data and initial set-up time. In analogy to event-day flowrate models developed here, a discrete simulation was performed with discrete output.

A major advantage of simulation models is the insight gathered from the shear detail and labor that goes into compiling a complex model application (Bedient et al, 1992). This implies that one will gain a fair amount of insight simply by reviewing the watershed in this manner. In its development the same was found to be true for flow regime modeling and associated instantaneous flowrate modeling conducted for Leading Creek. A distinct advantage of simulation models is that once set-up, they can be used to quickly and efficiently test various management schemes (e.g. reduced loading scenarios). This is not an attribute of flow regime and flowrate modeling conducted here. The trade off between the two is gained in readily identifiable trends in mass concentration across various flow regimes encountered and over time, greatly enhanced by fine spatial calibration. In this sense, complex and more simple simulations introduced here are both able to evaluate landscape-level impacts upon the quality and quantity of water (Bedient et al, 1992, Linsley et al, 1975). Uncertainty is more manageable in the latter.

Minimum flow/transport technical simulation model criteria (Bicknell, et al, 1993; USEPA, 1988, 1991) might include the following capabilities: a) ability to provide watershed-scale analysis of rainfall/run-off patterns for high, medium, and low flow stream conditions; b) ability to provide a calibrated flow model for use in sediment/contaminant fate and transport analysis; c) ability to identify significant point sources and non-point sources of sediment and contaminant loading in the stream system; and d) ability to evaluate reduced sediment and contaminant loading scenarios from sources identified. To integrate all but the last capability into an ecological risk assessment of Leading Creek, this study successfully pursued a simple course of separating water quality data and other stream flow-variant metrics into three classes describing low flow, medium flow, and high flow stream conditions. Ability was also developed to extend underlying data sets to evaluation of instantaneous flowrates across gaged and ungaged watersheds to compliment grab sampling and water quality analyses.

Flow models were applied here to water quality monitoring programs employing 1 to 2 day sampling events. The flowrate and flow regime modeling exercise evaluated in Leading Creek represents an intensive baseline methodology that might be extended to detailed calibration of multiple gaged sites in a formal simulation approach like HSPF. Data collection and analysis may also be able to evaluate uncertainty to a degree for ungaged sites in a more complex simulation. Combined flow regime and flowrate models developed in Leading Creek can both be viewed as discrete hydrologic simulations that employed a daily time step, integrated on a monthly basis, using stochastic and deterministic methods, respectively.

D.5.3 Integrating Multidisciplinary Team Hydraulic Monitoring

Differences in stage reading error rates between weekly monitoring and monthly monitoring were not anticipated. Coincidental synchronization on a daily basis, supplemented by continuous meters was though sufficient to characterize uncertainty in monthly stage data collected in conjunction with the water quality sampling program. Significant time, analysis, and in the end increased uncertainty was involved to address the issue. Due to this effort and the increased error induced, future designs would benefit with more purposeful synchronization of independent periodic monitoring. Confidence in final instantaneous flowrate modeling results developed for Leading Creek were in part the result of good fortune.

In comparison to monthly water quality sampling, weekly watershed head data had the distinct advantage of being collected on the same day within 2 to 2.5 hours at all sites, greatly reducing bias between stations. Timing becomes a more important factor in assessing or modeling more dynamic medium and high flow runoff conditions experienced within 24 hours of significant rainfall events. Evidenced by the exponent of flowrate models in Table D3, significant hydraulic variation between small and large subsheds occurred over the associated two-day sampling periods for medium and high flow conditions. Same day variation was often greatly reduced for similar events in widespread recession, for example as occurred in February

1997. Table D3 shows heterogeneity in rainfall is a challenging problem presented for any dynamic approach used to characterize hydrologic conditions. This applies equally to simplified approaches for ecological risk assessment and associated water quality monitoring programs as well as complex hydrologic simulation (Linsley et al, 1978). The four gages in Table D3 were positioned along a north-south trend perpendicular to the prevailing storm track pattern.

Using scientists and project technicians to collect head data at bridge reference points as they collect water quality data was complicated by several factors. This was a notable problem for monthly data collection using metal tapes. In retrospect, work plans should develop a detailed written instruction for this activity. Site maps should be included with statements on allowable tolerances, techniques to accurately collect measurements under normal and adverse conditions (e.g. wind), and contingencies if reference points have been disturbed. Electronic water level meters can greatly reduce measurement bias associated with visual estimation of the end of tape contact with the water surface. Used to address systematic and infrequent but large random errors, this work benefited significantly from concurrent weekly collections using an electronic meter that accompanied periodic monthly monitoring activities.

To assure accuracy, the electronic static water level meter technique should use two to three soundings with an up and down motion at the point close to mark. An electronic meter can be effectively substituted by careful measurement with a metal tape, weighted if necessary above the tape end. PVC pipes worked well as bridge reference points for multiple personnel use, but the advantage of conspicuousness made them susceptible to vandalism. It is recommended that staff gage design include 2 permanent, inconspicuous measuring points set on the bridge above the thalweg, and two reference control points set off the bridge. In addition to wind effects, visual inspection for surface turbulence should also be evaluated each time and noted in the record. Wind effects during weekly monitoring were observed infrequently for this work.

D.5.4 Errors in NPDES Flowrate Databases

In Figures D3 and D4, reported No. 31 daily discharge rates appeared to be underreported during some low and most medium flow condition events, with errors ranging up to 200%. With stage error reduced ten-fold and sampling rates increased ten-fold, on average errors increased for weekly data collected over the study period. This was evaluated by examination of direct measurement of flowrate at TS11 on Parker Run from weekly periodic monitoring, and comparing results to reported daily average levels of Mine No. 31 discharge rates. Comparison of two continuous mainstem gages, one above and the other below active Mine No. 31, supported this conclusion. Most notable were departures above expected mainstem trends at medium and low flow where active mine flows are significant with respect to natural instream flow. The two highest weekend data points shown in Figure D3 above the average flow condition value were also corroborated by trends in increased stage and specific conductivity. Because associated flow rates offset from the trend were extremely high with respect to reported maximum discharge rates, these two data points remain somewhat speculative.

Active mine permits required the collection of weekday data only. Elevated discharge patterns during weekends were sometimes also noted, though with less significance compared to magnitude in weekday reporting error. This was documented by the continuous gage just above Parker Run receiving Mine No. 31 flows. The CGS1 gage and Parker Run are subject to a minor, but observable slope-stage-discharge relationship under low to medium flows. Conclusions associated with incremental increases in Mine No. 31 discharge rates on weekends were based on observations of CGS1 level and an assumption that recessional flow should prevail during periods without significant rain or increased flow from Mine No 2. Low flow conditions associated with decreasing flow rates on the mainstem below the ASML mining belt would represent an offset to underreported active mine flows. This may explain convergence of CGS1 and CGS2 flowrates to expected average levels under those conditions seen in Figure D4, but not in Figure D3.

Based on rain, drought, and backwater data filters employed, Figure D4 does not represent matched weekend and weekday data. Differences in data on certain days shown in Figure D3 might in some cases be explained by potentially remaining run-off heterogeneity or lagged reporting by the mine. This would not address above average instream flow increases observed at TS11 though, or overall consistency of errors seen throughout the study. Undervalued weekday reporting could be easily explained by inaccurate slope or roughness used in pipe flow calculations associated with the primary measuring device.

The analysis here underscored key aspects of uncertainty associated with NPDES daily discharge data. The data are often used in mass balance calculations for watershed flow modeling or risk assessment. The need to verify accuracy of NPDES flow data, particular those based on manual and discrete interval measurements was evident in Leading Creek. Where NPDES permits do not require recordation of weekend flow rates, assumption of Friday's value may also introduce significant uncertainty in the flow modeling process, even where reported weekday variation appears minimal, the typical case for active Mine 31. In Leading Creek, routine flow reporting inaccuracy at active mines dominated unreported increases observed in weekend discharge rates. Though less significant here, overestimation can present similar concern in mass balance approaches using NPDES data

D.6 CONCLUSIONS

This paper focused on further development of a quantitative hydrologic-hydraulic analysis of Leading Creek to support quantitative-based risk assessment of the stream system. A previous evaluation was provided that described a methodology for flow regime modeling for use in ecological risk assessments. For application to accompanying water quality monitoring programs, work here extended that analysis for direct modeling and computation of instantaneous flowrates in ungaged subsheds, and for subsheds with missing data. By doing so, land use can now be brought into full focus by allowing for a detailed description and accurate analysis of watershed chemical loading rates normalized to unit drainage area. This paper described the technical basis underlying collection and use of hydrologic-hydraulic data, development of instantaneous flowrate models, and their intended application to water quality results for ecological risk assessment of Leading Creek.

The beneficial consequence of flowrate monitoring and modeling performed on Leading Creek was the developed ability to predict instantaneous flowrates and pollutant loading rates at 28 monitoring sites scattered throughout the watershed. Based on flow regime modeling concepts, this could be easily accomplished using head data from a single continuous gage when all sites exhibited relatively similar recessional hydrograph position. Variable runoff patterns can also be suitably determined for a variety of subshed sizes using multiple gages if they span sufficient ranges of drainage area. The multiple gages are needed to assess non-linearity and stability of the associated run-off pattern for a given sampling event. This work found that splitting 2-day sampling events by day was sufficient in most cases in maintaining stability and the ability to accurately describe non-linearity of the form $Qn = aA_d^b$. This work indicated that placement of a few selected gages encompassing small, medium and large subsheds in a watershed monitoring program can be effective in deriving useful event-day flow models for watershed wide application. The majority of these should be concentrated in smaller subsheds, where consideration of multiple gages in larger subsheds is more so guided by the need to assess variability in point source discharges present.

This work found that daily discharge rates from active underground mines appeared to be underreported during some low and most medium flow condition events, with errors ranging up to 200% of reported daily average values. With stage error reduced ten-fold and sampling rates increased ten-fold, on average errors increased for weekly data collected over the study period compared to monthly sampling data. This was evaluated by examination of direct measurement of flowrate in the receiving stream from weekly periodic monitoring, and comparing results to reported daily average discharge rates in USEPA's NPDES Mor database. Comparison of two continuous mainstem gages, one above and the other below the active mine supported this conclusion. Though less minor with respect to overall underreporting errors, this work also found that uncertainty in weekend NPDES data may be increased for operations where only weekday reporting is required. Assumption of Friday data for weekends in time-sequence data management can lead to significant error, even when little weekday variation is noted. Neither of these aspects may typically be suspected as sources of uncertainty in watershed modeling programs but need to be addressed.

Electronic water level meters for use in stage measurement in conjunction with well controlled vertical benchmarks on bridgedecks greatly reduced measurement bias in stream flow measurement programs attempted. Significant systematic bias was identified with visual estimation of the end of tape contact with the water surface using metal tapes, resulting in over measurement of stage and underestimation of rated flows. Use of the metal tapes requires a great deal of care in measurement technique. Used to address systematic and infrequent but large random errors, this work benefited significantly from concurrent weekly collections using an electronic meter that accompanied periodic monthly and continuous stage monitoring activities. Continuous mainstem meters were also critical to characterizing watershed flowrates and in assessing daily stability in rainfall, run-off, and NPDES discharges in the watershed.

The hydrologic-hydraulic modeling framework proposed here supported successful classification of pollutant mass concentration and loading rates for comparison to aquatic criteria and quantified unit-area land use, respectively. Expectedly, site-specific evaluations of flowrate as opposed to single gage assignment allowed far more accurate descriptions of flow regime for assessment of both mass concentration and unit-area loading. The degree of improvement depended upon temporal and spatial variation in water flow and landscape pollutant loading experienced, where benefits decrease with increasing water quality sampling frequency.

D.7 LITERATURE CITED

See Chapter 2 for Literature Cited

Appendix E - Staff Gage Vertical Elevation Surveys

Justin Eric Babendreier

Table E1 - Staff Gage Station Survey
Initial Inverted Staff Gage Control Survey - Leading Creek SGS Stations Referenced To Assumed Benchmark

Weather/Air Temp: Sunny to Partly Cloudy, 40-50 °F 3/26-27/96, Overcast 35-40 °F 3/29/96
Water Level Data Collected By: D. Wright, J. Babendreier
Survey Data Collected By: D. Wright, J. Babendreier 3/27/96; J. Babendreier, M. Yeager, R. Currie, D.S. Cherry 3/29/96
Equipment Used : Solinst 100' Static Water Level Meter w/ Env. Probe, Autolevel Pentax Pal - 5C SN # 52143-1 ;13 ft wooden rod

** All data given in feet; original units of measuremnt*

Staff Gage ID / No.	Station ID / No.	Station Name/ Location	Measuring Point Gage Install Time	Measuring Point Gage Install Date	Depth to Water from PVC MP	Spad Survey Accuracy Bridge/Tree	Survey Date	Bridge Spad BM Elevation	Tree Spad Relative Elevation	MP Relative Elevation
SGS1	LCS1	@ Albany	10:10	3/26/96	12.18	0.01 to 0.02	3/29/96	0	1.90	2.87
SGS2	TS3A	Fivemile Run	10:48	3/26/96	9.85	0.01 to 0.02	3/29/96	0	3.11	4.00
SGS3	LCS2	@ Carpenter	11:28	3/26/96	15.87	0.01 to 0.02	3/29/96	0	-0.43	3.86
SGS4	LCS3	@ Sisson	17:35	5/16/96	11.03	MP = 0.21 feet above top edge of guard rail post supporting MP				
SGS5	LCS4	@ Dyesville	11:50	3/26/96	11.34	0.01 to 0.02	3/29/96	0	0.58	2.39
SGS6	TS8B	Mud Fork	12:14	3/26/96	9.74	0.01 to 0.02	3/29/96	0	1.60	2.89
SGS7	LCS5B/CGS1	@ Dexter South	12:30	3/26/96	17.46	0.01 to 0.02	3/26/96	0	7.87	2.66
SGS8	TS11	Parker Run	12:46	3/26/96	8.15	0.01 to 0.02	3/26/96	0	0.34	-0.28
SGS9	LCS6	@ Malloons	13:05	3/26/96	16.05	0.01 to 0.02	3/26/96	0	0.71	2.77
SGS10	LCS7	@ Corn Hollow	13:32	3/26/96	16.27	0.01 to 0.02	3/26/96	0	1.50	2.80
SGS11	TS14	Little Leading Creek	13:54	3/26/96	11.63	0.01 to 0.02	3/26/96	0	2.19	3.17
SGS12	LCS8	@ Twin Bridges	14:15	3/26/96	20.85	0.01 to 0.02	3/26/96	0	1.33	2.12
SGS13	TS16	Paulins Hill Run	14:45	3/27/96	3.14	0.01 to 0.02	3/26/96	0	6.48	-0.47
SGS14	LCS9/CGS2	@ Iron Bridge	14:48	3/26/96	18.92	0.01 to 0.02	3/26/96	0	1.40	3.92
SGS15	TS17	Thomas Fork	15:07	3/26/96	17.50	0.01 to 0.02	3/26/96	0	1.58	2.71
SGS16	LCS10	@ Middleport	15:46	3/26/96	25.79	0.01 to 0.02	3/26/96	0	1.67	3.91

Notes:
MP = Measuring Point
LCS4 and LCS10 original pole-tree spads may be incorrectly reported 1.0 units too high based on differences noted in two independent surveys.
Corrected by 1.0 units, tree-pole values at LCS4 and LCS10 would solve errors found between 2 surveys; true also for new benchmark surveys corrected in same manner.

Table E2 - Staff Gage Station Survey
Controlled Benchmark Vertical Elevation Survey - Leading Creek

Survey Data Collected By: **AEP - 5/97; performed by local professional land surveyors**
Equipment Used : **Dual GPS Receiver or Control Traverse From USGS Benchmarks Referenced to Feet Above Mean Sea Level (AMSL).**
Surrogate measuring point (MP) indicates a field mark on the bridge structure has replaced the original PVC pipe gage installed in 3/96, usually due to vandalism or construction.
Purpose of this analysis is to verify on-site benchmark control for elevation and stream slope models and to verify accuracy of surrogate measuring point control at altered sites.

** All data given in feet; original units of measuremnt*

					Measured During Second Survey					
Station ID / No.	**Station Name/ Location**	**Benchmark Control Comment**	**Benchmark Est. Accuracy Vertical (ft)**	**New Bridge Ref. Spad Installed 5/97**	**Orig. Spad in Bridgedeck Elev. (feet)**	**Orig. Spad in Pole (Tree) Elev. (feet)**	**Top of Plastic MP Elev. (feet)**	**Top of Surrogate MP Elev. (feet)**	**Date Surrogate MP Started**	**Surrogate MP Location Description**
LCS1	@ Albany	GPS	0.1 to 1.0	-	681.93	683.88	Missing	684.21	3/31/97	Top of guardrail post
TS3A	Fivemile Run	GPS	0.1 to1.0	-	637.02	640.00	641.04			-
LCS2	@ Carpenter	Traverse Point	0.1	Lost spad	630.13	628.82	633.16			-
LCS3	@ Sisson	GPS	0.1 to1.0	New survey	617.31	619.83	620.04			-
LCS4	@ Dyesville	GPS	0.1 to1.0	Resurfaced	601.22	602.01	604.83			-
TS8B	Mud Fork	C&GS BM H224	0.1		596.12	597.67	Missing	598.48	6/20/96	Top of guardrail
LCS5B/CGS1	@ Dexter South	GPS	0.1 to1.0	-	595.56	603.46	598.18			-
TS11	Parker Run	Traverse Point	0.1	-	604.61	604.87	604.31			-
LCS6	@ Malloons	C&GS BM C224	0.1	-	588.88	589.21	Missing	591.24	6/26/96	Top of guardrail
LCS7	@ Corn Hollow	C&GS BM Y207	0.1	-	574.53	575.93	577.36			-
TS14	Little Leading Creek	GPS	0.1 to1.0	Guardrail post	Missing	573.09	574.06			No new bridgedeck spad set
LCS8	@ Twin Bridges	GPS	0.1 to1.0	Resurfaced	571.20	572.66	Missing	573.22	11/19/96	Top of guardrail
TS16	Paulins Hill Run	GPS	0.1 to1.0	-	580.28	586.56	579.80	579.63	7/18/97	Top of conc. wingwall, downstream
LCS9/CGS2	@ Iron Bridge	GPS	0.1 to1.0	-	560.20	Not surveyed	564.06			Culvert = orig. tree-pole spad
TS17	Thomas Fork	GPS	0.1 to1.0	-	560.23	561.84	Missing	562.32	1/28/97	Top of guardrail
LCS10	@ Middleport	GPS	0.1 to1.0	New spad set	565.44	566.29	Missing	569.13	5/19/96	Top of vertical rail support

Notes:
LCS10 measuring point (MP) missing and bridge spad reset. Based on backwater-pool data two days before and two days after 5/19/96, surrogate MP was 0.41 ft lower than original.
LCS6 distance between guardrail and original pipe MP = 16.14-15.775 = 0.365ft, measured 5/16/96 during profiling. Surrogate corrections track profiles before and after.
TS8B 5/19/96 cross-section profile 0.55' delta in surrogate. Guardrail and bridgedeck elevations inadvertently switched during recording, guardrail should be higher elevation.
TS16 surrogates assumed same elevation top of culvert both sides of road. See notes for temporary surrogate used for thee weeks in late July and August 1997.
At TS17, backwater-pool data at LCS9 and LCS10 across three events before change and three events after indicated surrogate shift at TS17 was 0.49 ft.
LCS1and LCS8 original MP datum calculated from 1st survey. Surrogate differences are consistent from 1st survey where photos also indicated table values are reasonable.
LCS4 and LCS10 new pole-tree spad survey data may be incorrectly reported 1.0 units to high based on differences noted in two independent surveys.
Corrected by 1.0 units, tree-pole values at LCS04 and LCS10 would solve errors found between surveys; would also be true if corrected original surveys in similar manner.

Table E3 - Staff Gage Station Survey
Error Analysis for Onsite Vertical Elevation Control Points

Survey Data Collected By: AEP - May 1997 ; AEP and VA Tech - March 1996

Equipment Used : Survey #2 - GPS or Control Traverse From USGS Benchmarks, Survey #1: Autolevel & Rod with assumed benchmark control (e.g. bridgedeck spads)

Surrogate measuring point (MP) indicates a field mark on the bridge structure has replaced the original PVC pipe gage installed in 3/96, usually due to vandalism or construction.
Purpose of this analysis is to verify on-site benchmark control for elevation and stream slope models and to verify accuracy of surrogate measuring point control at altered sites.

** Data shown relative to survey control benchmark datum = bridgedeck spad. All data given in feet; original units of measuremnt*

		Survey #2 Results						Survey #1 Results			Comparison Between Surveys		
Station ID / No.	**Station Name/ Location**	**Bridgedeck Spad-Datum Elevation**	**Spad in Pole (Tree) Elev.**	**Top of Pipe MP Elev.**	**Top of MP Surrogate Elev.**	**Orig. Top Pipe MP Elev.**	**Surrogate MP Offset Elev.**	**Orig. Pole Spad Elev.**	**Original Pipe MP Elev.**	**New Bridge Spad**	**Bridge-MP Onsite Elev. Error**	**Bridge-Pole Onsite Elev. Error**	**Pole-MP Onsite Elev. Error**
LCS1	@ Albany	681.93	1.95	Missing	2.28	2.87	0.59	1.90	2.87	No	**N/A (0)**	0.05	0.05
TS3A	Fivemile Run	637.02	2.98	4.02				3.11	4.00	No	**0.02**	-0.13	-0.15
LCS2	@ Carpenter	630.13	-1.31	3.03				-0.43	3.86	Yes	-0.83	-0.88	**-0.05**
LCS3	@ Sisson	617.31	2.52	2.73				0	0.21	-	-	-	**0.00**
LCS4	@ Dyesville	601.22	0.79	3.61				0.58	2.39	Yes	1.22	0.21	**-1.01**
TS8B	Mud Fork	596.12	1.55	Missing	2.36	2.89	0.53	1.60	2.89	No	**0.00**	-0.05	-0.05
LCS5B/CGS1	@ Dexter South	595.56	7.90	2.62				7.87	2.66	No	**-0.04**	0.03	0.07
TS11	Parker Run	604.61	0.26	-0.30				0.34	-0.28	No	**-0.02**	-0.08	-0.06
LCS6	@ Malloons	588.88	0.33	Missing	2.36	2.73	0.37	0.71	2.77	No	**-0.04**	-0.38	-0.34
LCS7	@ Corn Hollow	574.53	1.40	2.83				1.50	2.80	No	**0.03**	-0.10	-0.13
TS14	Little Leading Crk.	Missing	573.09	574.06				2.19	3.17	Not set	**N/A**	N/A	0.01
LCS8	@ Twin Bridges	571.20	1.46	Missing	2.02	2.12	0.10	1.33	2.12	Yes	**N/A (0)**	0.13	0.13
TS16	Paulins Hill Run	580.28	6.28	-0.48	-0.65		0.17	6.48	-0.47	No	-0.18	-0.20	**-0.02**
LCS9/CGS2	@ Iron Bridge	560.20	No survey	3.86				1.40	3.92	No	**-0.06**	N/A	N/A
TS17	Thomas Fork	560.23	1.61	Missing	2.09	2.58	0.49	1.58	2.71	No	**-0.13**	0.03	0.16
LCS10	@ Middleport	565.44	0.85	Missing	3.69	4.10	0.41	1.67	3.91	Yes	0.19	-0.82	**-1.01**

Notes:

The method compares the results of the vertical benchmark survey against the original 3/96 autolevel/rod survey used to establish onsite survey control at each staff gage site
Relative locations between control points and measuring points (MPs) from survey # 1 are considered accurate to +/- 0.01 to 0.02 ft, though GPS errors may range 0.1' to 1'[+].
Original MP elevation was calculated from 1st survey if pipe missing; LCS1 & LCS8 corroborated by photo and LCS10, LCS6, TS8B, TS16 & TS17 verified from other field data.
N/A indicates one or more control point(s) was lost and therefore no comparison can be made (e.g. new spad or MP installed between surveys). N/A(0) indicates zero by assumption.
For interpretation, as an example, large similar errors between bridgedeck and other points with small tree/pole-MP errors indicates effect of new spad placement in bridegdecks.
LCS4 and LCS10 original or new benchmark pole-tree spads may be incorrectly reported 1.0 units based on differences noted between the two independent surveys.
Errors between on-site points of 0.1' up to 1.0' are possibly attributed to drift and offset error in GPS receiver control; bridge spads and MPs typically were in close horizontal proximity.
Tree/pole spads represent survey nail tips set perpendicular to gravity and may explain some errors observed between surveys. Most spads set with nails and shiners, 2 set with chisel marks.

Table E4 - Staff Gage Station Survey
Station Elevation Survey Summary - Leading Creek

Surrogate measuring point (MP) indicates a field mark on the bridge structure has replaced the original PVC pipe usually due to vandalism or construction.

** All data given in feet; original units of measuremnt*

Station ID / No.	Surr. MP Date	Bridge Deck Elev.	Tree/ Pole Elev.	MP Elev.	Surr. MP Elev.	Surr. MP Offset	BM Survey Offset	Bridge Datum (DTW)	Surr. Datum (DTW)	Staff Gage Datum	Offset Gage Datum	BM Elev. Error	Surr. MP Location
LCS1	3/31/97	681.93	683.88	684.80	684.21	0.59	-	15	14.41	669.80	669.80	1	Top of Guard. Post
TS3A		637.02	640.00	641.04	641.04	0	-	12	12	629.04	629.04	1	
LCS2		630.13	628.82	633.16	633.16	0	-	20	20	613.16	613.16	0.1	
LCS3		617.31	619.83	620.04	620.04	0	-	16.5	16.5	603.54	603.54	1	
LCS4		601.22	602.01	604.83	604.83	0	-	15.5	15.5	589.33	589.33	1	
TS8B	6/20/96	596.12	597.67	599.01	598.48	0.53	-	13.53	13	585.48	585.48	0.1	Top of Guardrail
LCS5B/CGS1		595.56	603.46	598.18	598.18	0	-	24.33	24.33	573.85	573.85	1	
TS11		604.61	604.87	604.31	604.31	0	-	12	12	592.31	592.31	0.1	
LCS6	6/26/96	588.88	589.21	591.61	591.24	0.37	-	20.37	20.00	571.24	571.24	0.1	Top of Guardrail
LCS7		574.53	575.93	577.36	577.36	0	~0	22	22	555.36	555.36	0.1	
TS14		-	573.09	574.06	574.06	0	-	15	15	559.06	559.06	1	
LCS8	11/19/96	571.20	572.66	573.32	573.22	0.10	~0	26	25.9	547.32	547.32	0.1	Top of Guardrail
TS16	7/18/97	580.28	586.56	579.80	579.63	0.17	-	5	4.83	574.80	574.80	1	Top of Culvert
LCS9/CGS2		560.20	-	564.06	564.06	0	0.95	25.39	25.39	538.67	537.72	0.1	
TS17	1/28/97	560.23	561.84	562.81	562.32	0.49	-1.57	21	20.51	541.81	543.38	0.1	Top of Guardrail
LCS10	5/19/96	565.44	566.29	569.54	569.13	0.41	~0	34.41	34	535.13	535.13	0.1	Top of Rail Support

Notes:

MP = Measuring point - all originally constructed from 2" PVC pipes rigidly attached to bridge structure.

DTW = Depth to water; Surr. = Surrogate; BM = Benchmark.

Bridge datum values were assumed and roughly approximate the minimum bed elevation for the reach. Stations LCS5B and LCS9 established by aspects of CGS installation.

Benchmark survey offset represents station to station calibration data determined via analysis of static pools under backwater flood conditions.

Benchmark elevation error stated incorporated adjustments for pool calibration of stations TS17, LCS8, LCS9, and LCS10.

Relative stage was determined by subtracting static water level measurement (DTW) from bridge datum or surrogate bridge datum value, depending on surrogate date.

Elevation above mean seal level was determined by adding relative stage to staff gage datum, adjusted for pool calibration offsets at LCS9 and TS17.

During extreme low flow conditions, occasional use was made of alternate points on guard rails where data was adjusted manually to the applicable datum above.

At some sites, occasional maintenance was required to clear sediment build up directly below the MP.

Table E5 - Leading Creek Benchmark Survey Calibration Using Flood Pools Induced by Ohio River; Represents 1996 to 1998 data referenced to surveyed benchmark elevations. Correction for head $h_{ij}=0$ was given by average offset values measured for LCS9 and TS17. Offsets for LCS7-LCS8 and LCS8-LCS10 were attributed to measurement error or slopes >0.

Gage Relation	Head Avg. (m)	Head σ (m)	Reach Length (km)	Pool Events -n-
LCS7-LCS8	0.044	-	5.2	1
LCS8-LCS10	-0.009	0.008	9.6	2
LCS9-LCS10	0.289	0.017	3.4	8
T17-LCS10	-0.482	0.013	2.2	9
T17-LCS9	-0.768	0.014	5.2	8
T17-LCS8	-0.483	0.015	11.4	2

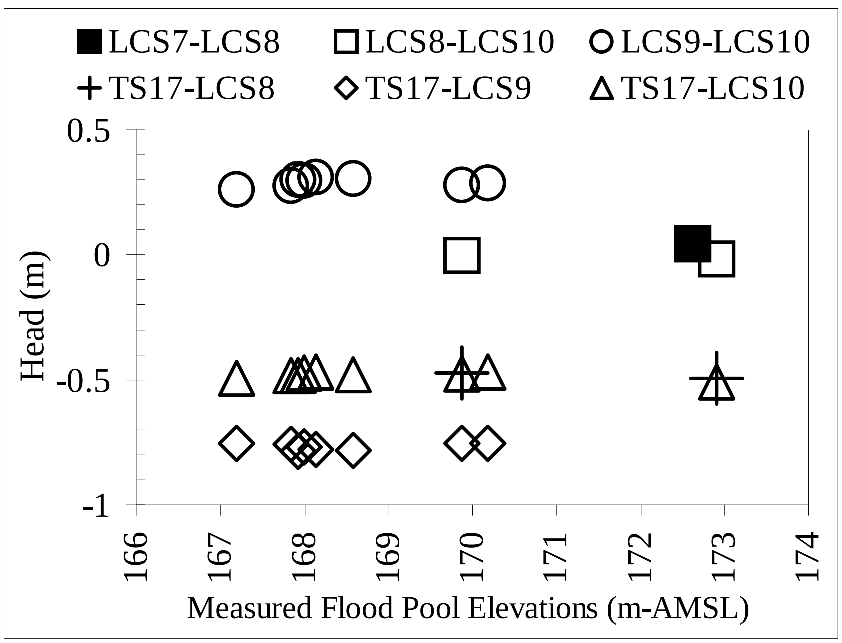


Figure E1 - Leading Creek Benchmark Survey Offsets Measured During Ohio River Backwater Pool Conditions; Referenced to benchmark survey elevations, data was based on gage relations for 1996 to 1998. Benchmark calibration solving for $h_{ij}=0$ (+/- 3 to 30 mm) was given by pool average head offset values measured for LCS9 and TS17. Pool analysis indicated reported onsite GPS tolerance control (+/- 0.3m) was underestimated by 50% at station TS17.

Benchmark Survey Analysis

Except for two tree-spad control point corrections of exactly 1 unit at stations LCS10 and LCS4 (1 ft. = 0.3m), error tracking between two elevation surveys was good. There was evident sometimes more error associated with tree spads. All bridgedeck measuring point (MP) comparisons showed small errors typically (<15 mm), where new bridgedeck spads showed consistent errors across tree-pole spads and pipe spads, and small relative pole-pipe errors. Final analysis confirmed all staff gauge survey datum were accurate. For March 1997, the crest pool flood mark was determined from the benchmark survey via TS17 flood mark at 172.48 m AMSL, less a 60 mm bolt offset and absolute errors used to offset pool corrections. This gave a final pool crest value of 567.25 feet AMSL or 172.90 m AMSL for the morning of 3/4/97. Data clearly shows that upper station LCS7 did not reach near-pool levels, an extremely rare event, until 3/5/97, a day after the pool reached LCS8. For the flood of 1997, the estimated time of peak at non-accessible sites was based on similar time of travel analysis using flood waves and kinematic wave velocity measured in June 1996.

A review of pool events at the creek mouth across two years indicated that the TS17 surrogate measuring point shifted ~ 0.16 m and not 0.19 m calculated via the surveys. Of sites with surrogate measuring points induced by vandalism, TS17 was the only site without independent reference data to check accuracy of on-site GPS control and possible drift. Pools for 12/4/97, the 5/15-19/96 sequence, 3/12/97, 1/13/98, 3/23/98, and 4/23/98 all ranged 0.759 m to 0.780 m, centered about 0.771 m head change at TS17 to LCS9, where LCS9 was not vandalized. Similar ranges of error were also estimated from head readings between T17 and LCS10. Surrogate control point corrections were made prior to pool calibration. Again due to vandalism, the LCS10 surrogate measuring point offset checked via pool analysis and survey control was corrected 0.12 m. Enclosed survey tables and notes document other surrogate control data and associated comments.

As indicated the TS17 surrogate was found to be in error by 30 mm, based on initial survey data. Associated TS17, LCS1 and TS8B stage-discharge profiles were corrected after the survey was finalized. Thus all velocity profiles used in stage-discharge curves reflected an accurate, consistent datum. The surrogate checks and error analysis between surveys showed good results except for the noted 0.3m offsets of tree-spads at LCS10 and LCS4. Surrogate control point changes were checked with two independent data sets and appear representative. Inconclusive in directly establishing a measure of surrogate offset, independent data was consistent with final analysis reviewed for calculated surrogate MPs at TS17 and LCS8. TS16 surrogate data was originally calculated incorrectly since the pipe gauge was below top-o-culvert (Cherry et al, 1999). Final analysis assumed the surrogate culvert elevation was 76 mm higher than the original top-o-pipe. This did not affect stage ratings completed before the surrogate date. Top-o-pipe was 85 mm above the culvert headwall, and culvert top surface from photographic evidence was assumed equivalent to the road spad. LCS9 stage was also incorrectly overstated by 85 mm for the 1997 flood crest pool calculation at 176.46 m AMSL corrected stage for this and a subsequent 0.15 m drop (Cherry et al, 1999).

Appendix F - Stream Elevation and Slope Models

Justin Eric Babendreier

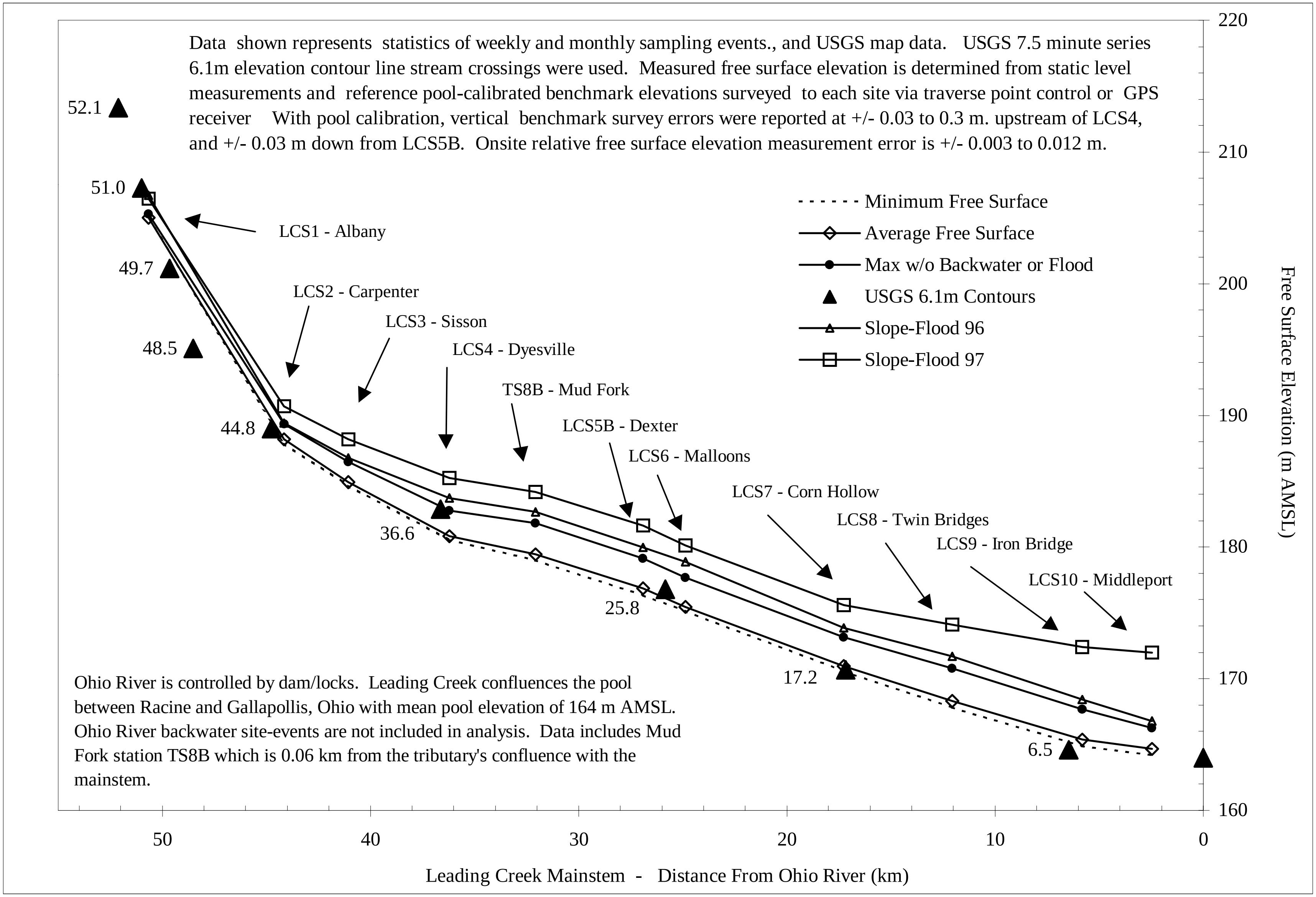


Figure F1 – Leading Creek Free Surface Elevation Analysis Based On Periodic Head Measurements

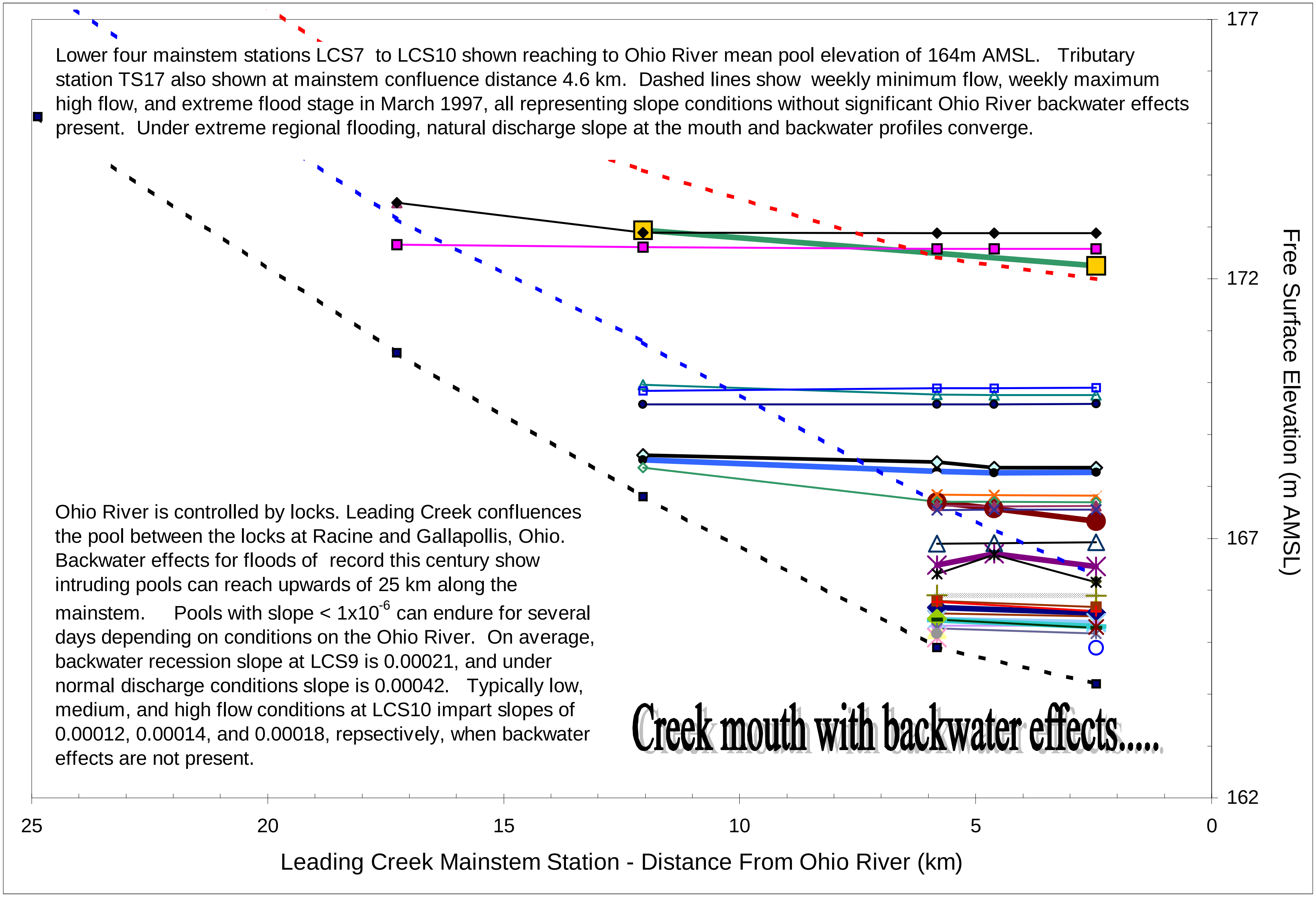


Figure F2 – Leading Creek Mouth Backwater Analysis Based On Periodic Head Measurements

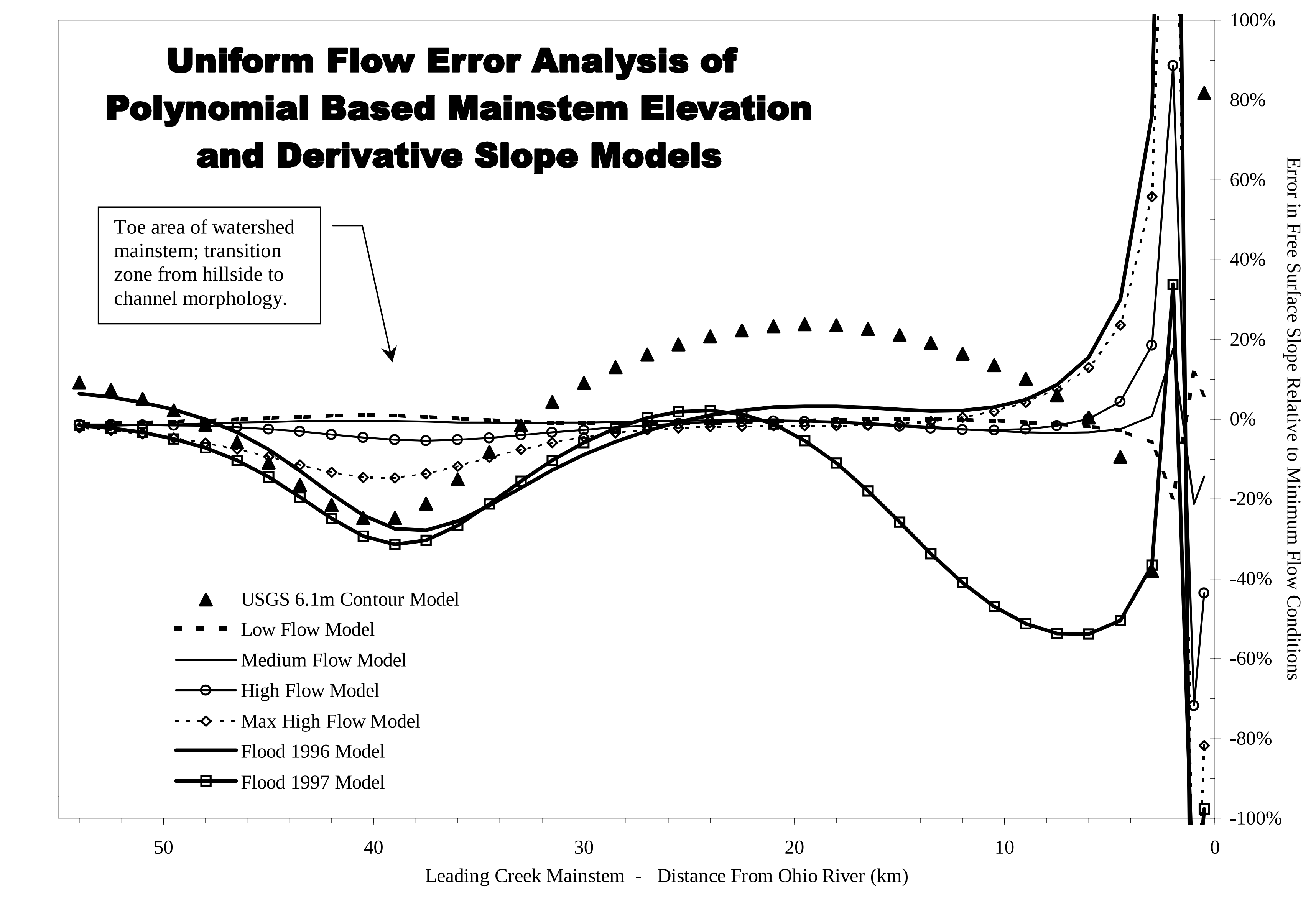


Figure F3 – Leading Creek Free Surface Slope Error Analysis With Respect to Minimum Flow Condition Model Results

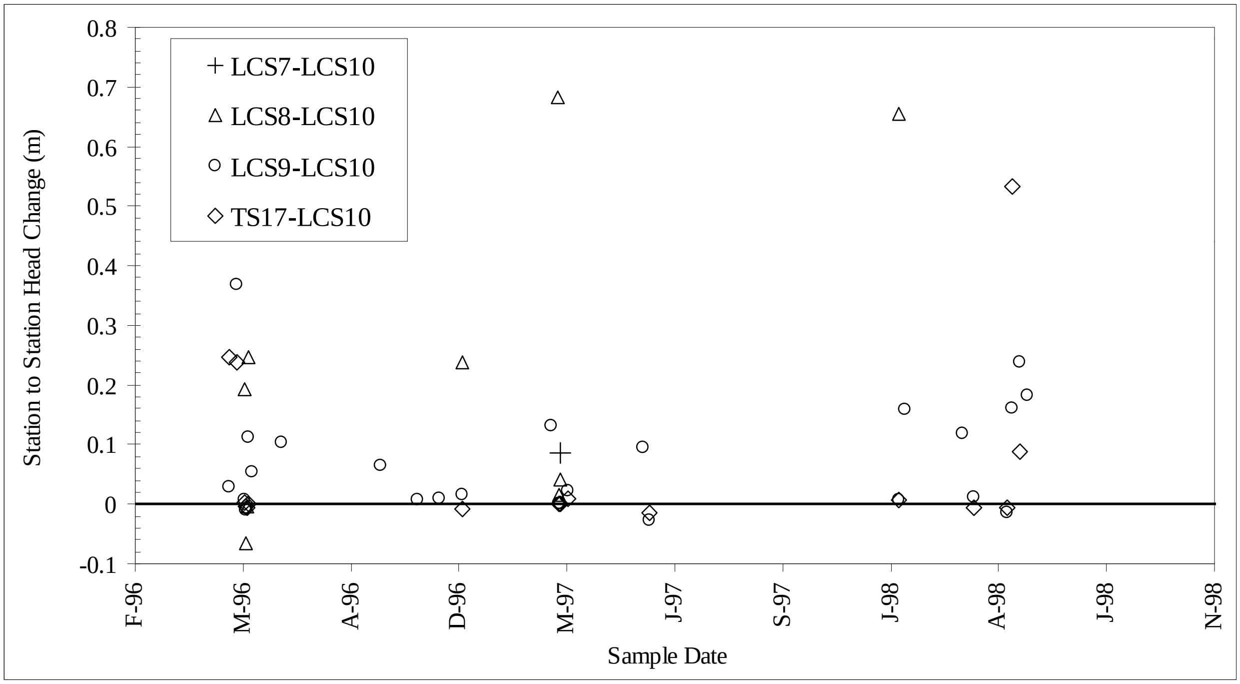


Figure F4 – Leading Creek Mouth Backwater Conditions (After Ohio River Pool Calibration)

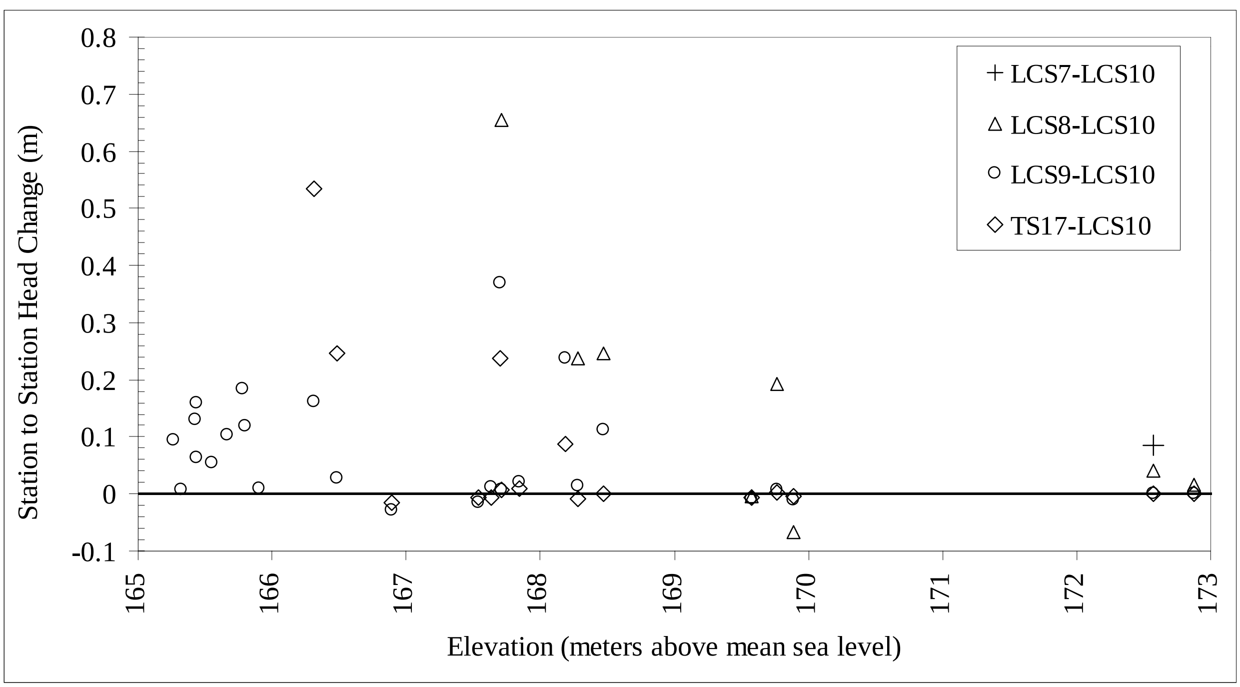


Figure F5 – Leading Creek Mouth Backwater Conditions Relative to Station LCS9

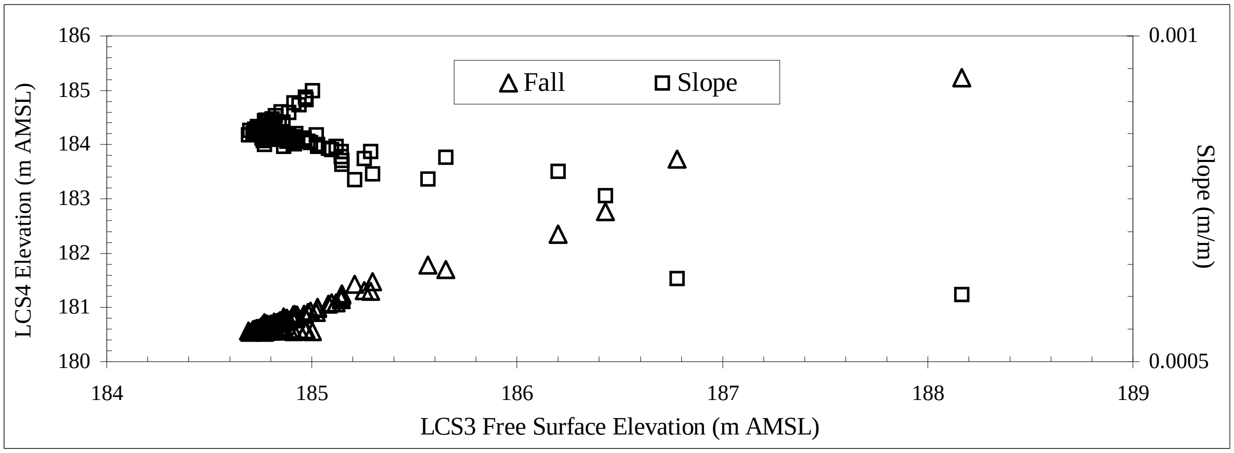


Figure F6 – Free Surface Fall Analysis – Stations LCS3 and LCS4. Shows periodic beaver dam influence at LCS3 where otherwise relatively constant slope prevails during non-flood conditions.

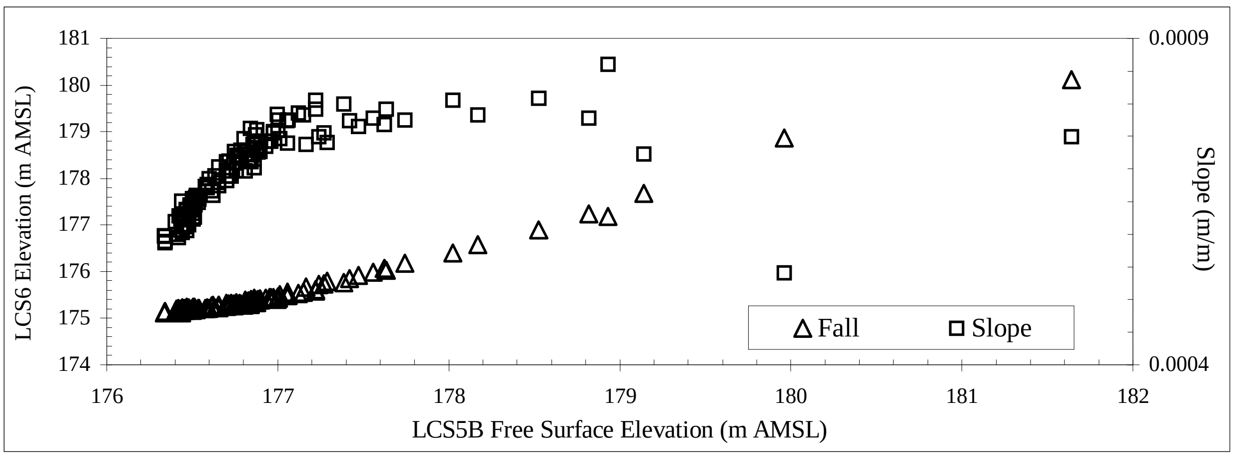


Figure F7 – Free Surface Fall Analysis – Stations LCS5B and LCS6. Natural non-uniform flow can occur in transition from low to medium regimes. Shows how elevation and slope models based on averaged flow conditions can dampen true slope over short distances.

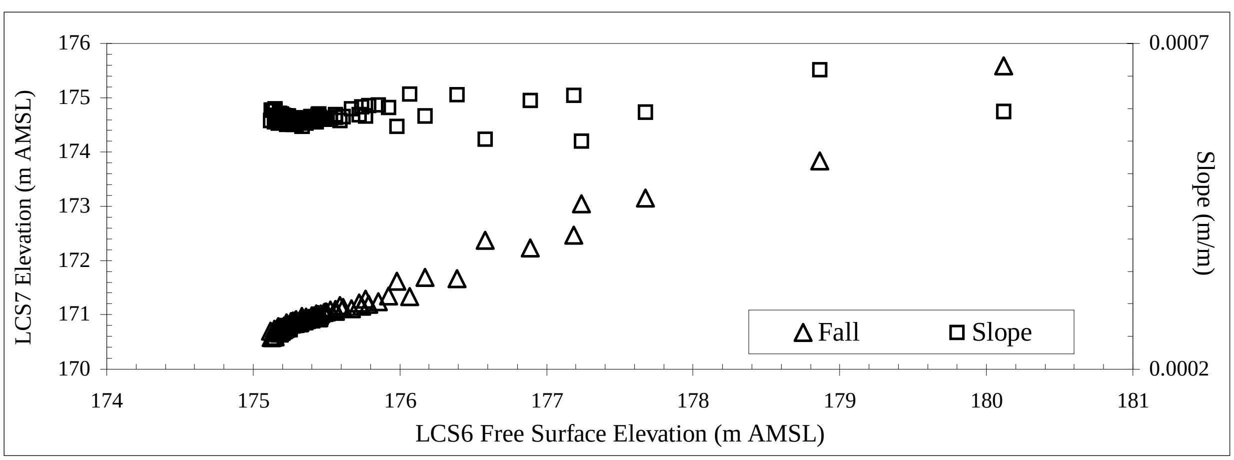


Figure F8 – Free Surface Fall Analysis – Stations LCS6 and LCS7. Shows relatively constant slope characteristic of most gage stations investigated, supporting assumptions for uniform flow.

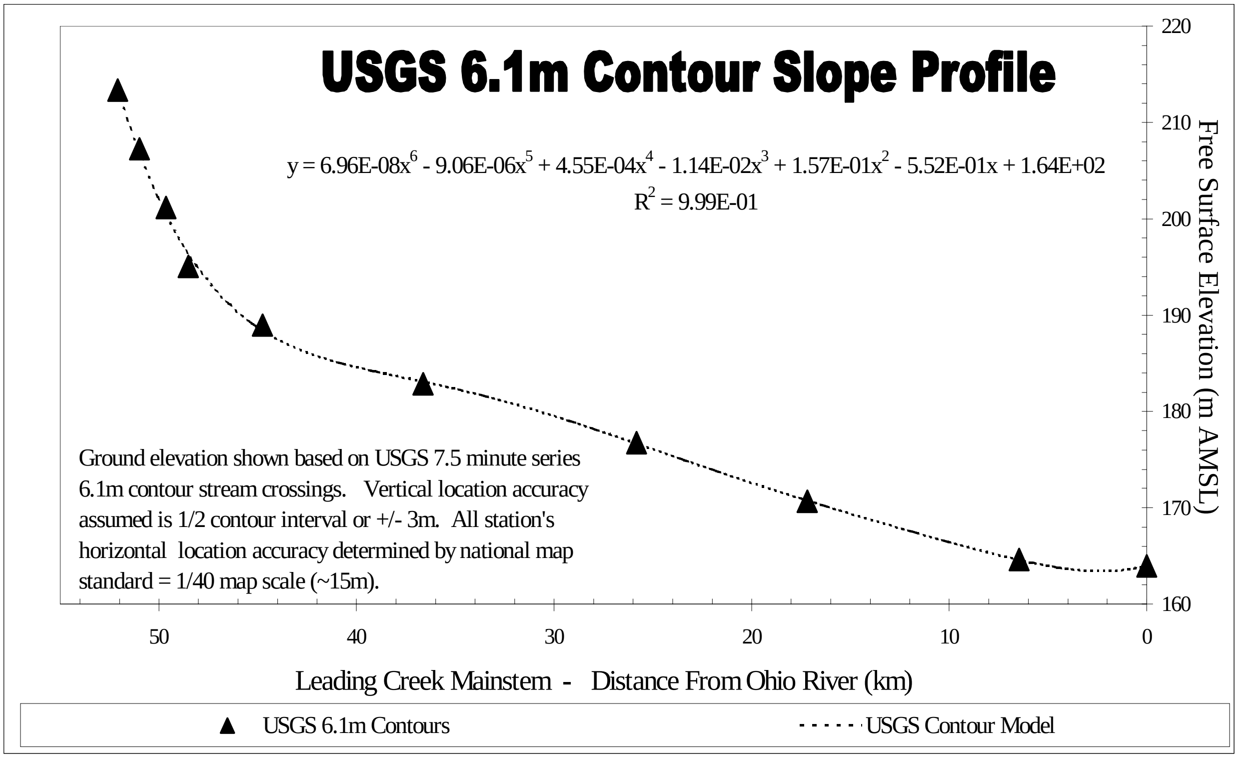


Figure F9 – Leading Creek USGS 6.1m Contour Elevation Model

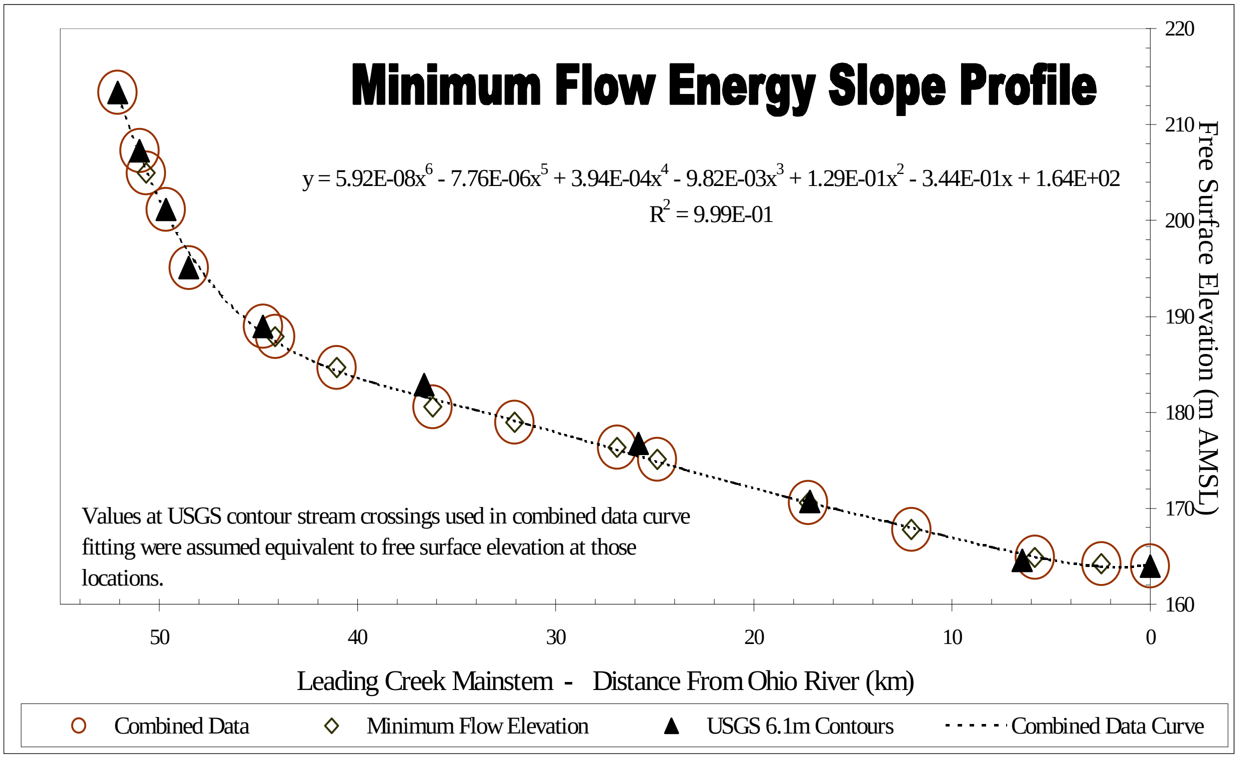


Figure F10 – Leading Creek Minimum Flow Condition Elevation Model

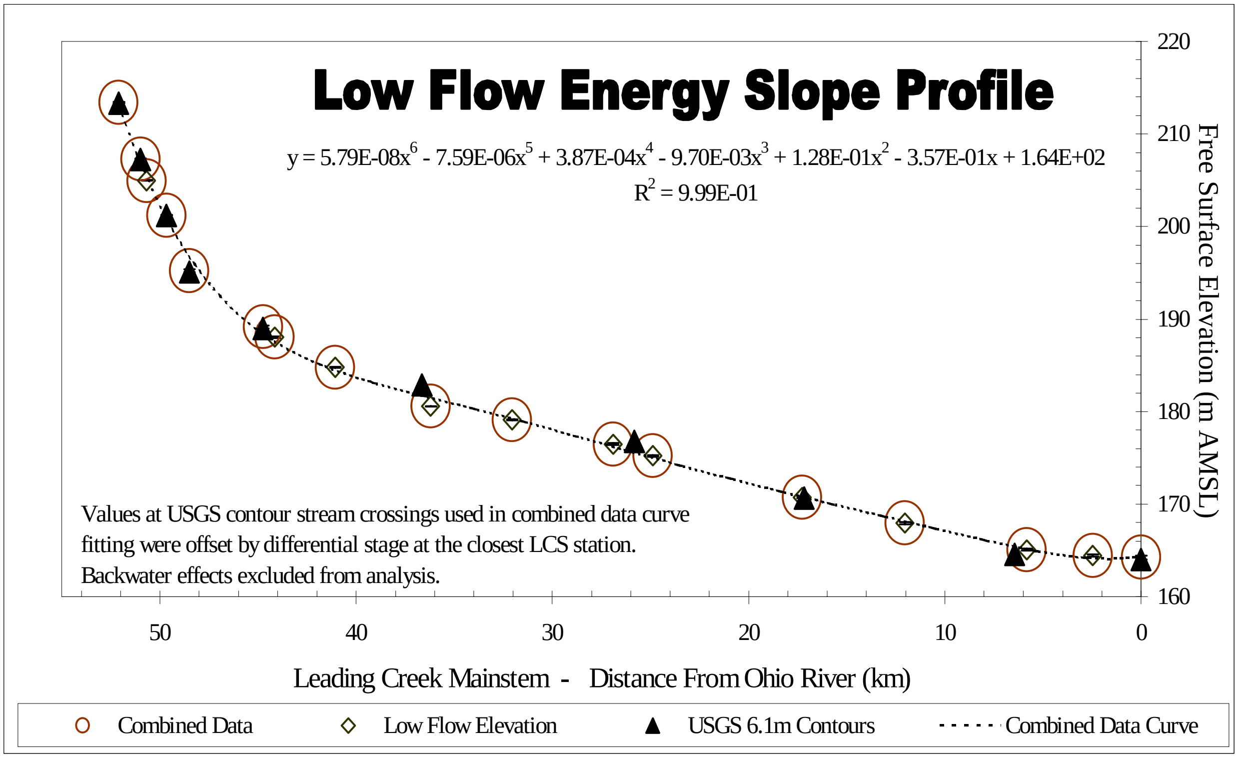


Figure F11 – Leading Creek (Mean) Low Flow Condition Elevation Model

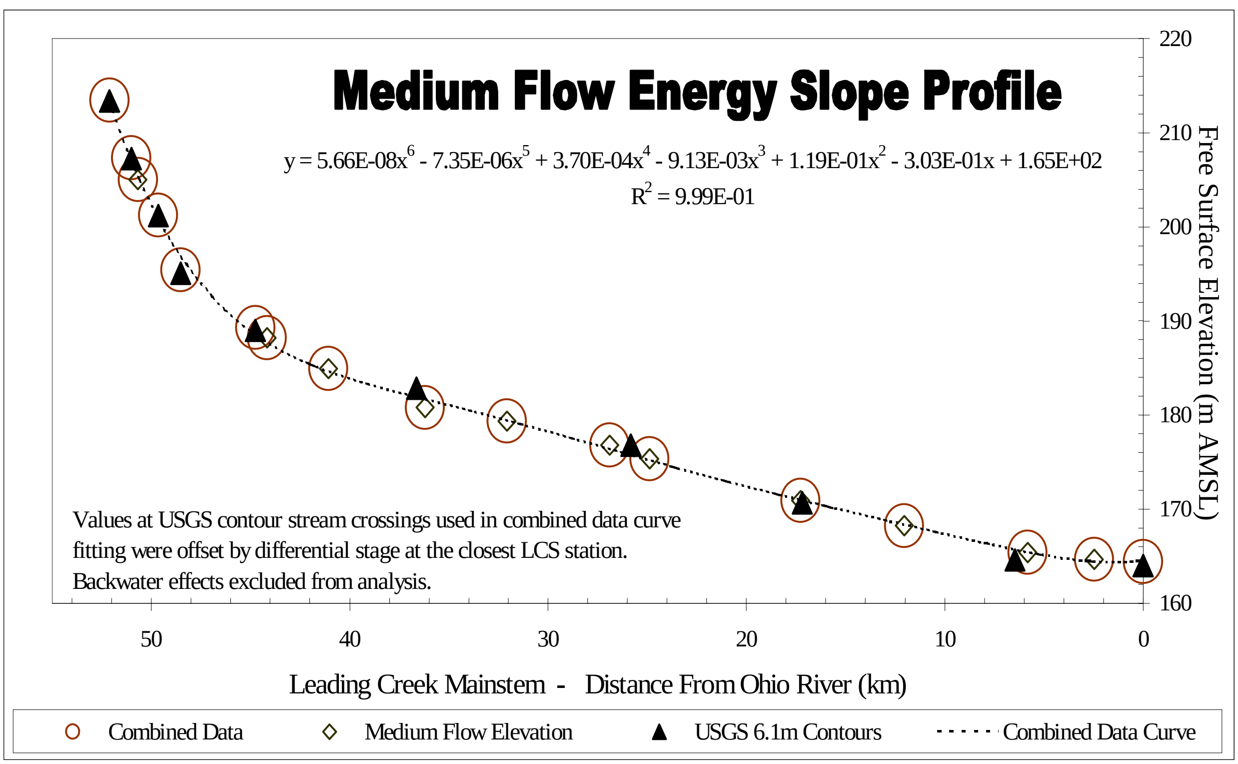


Figure F12 – Leading Creek (Mean) Medium Flow Condition Elevation Model

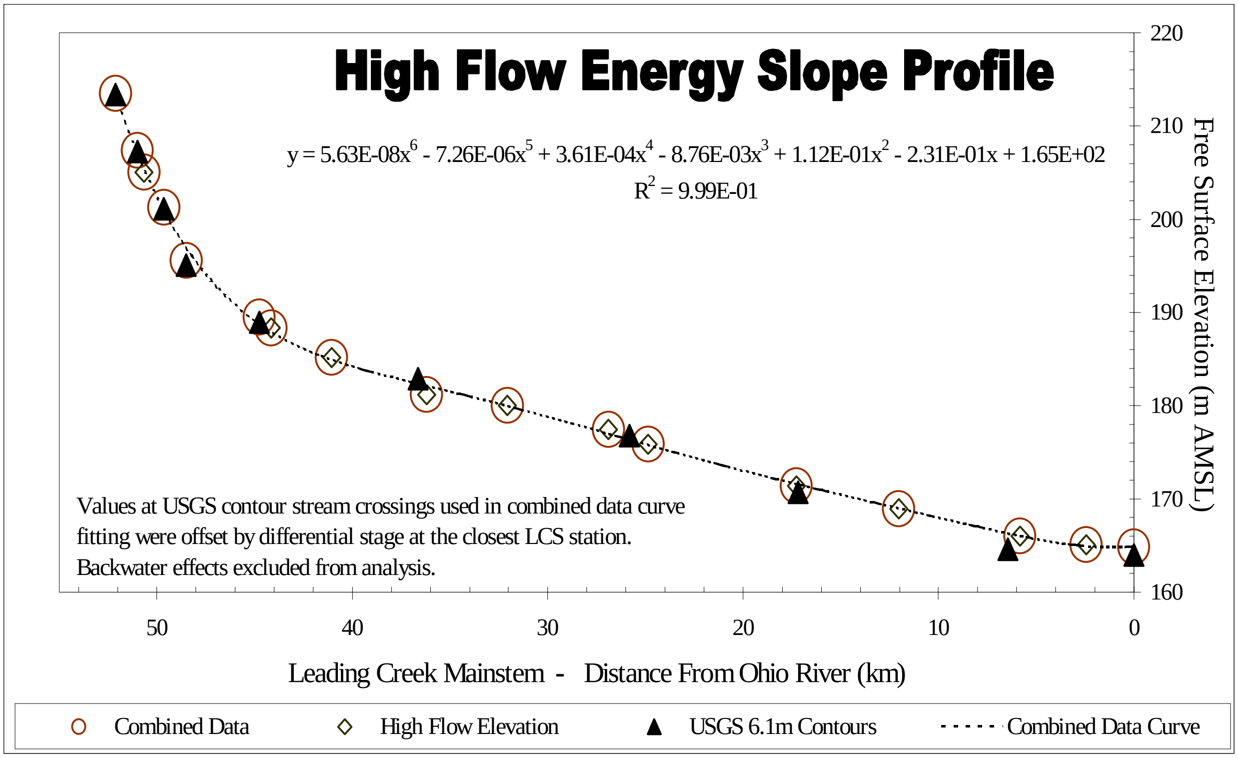


Figure F13 – Leading Creek (Mean) High Flow Condition Elevation Model

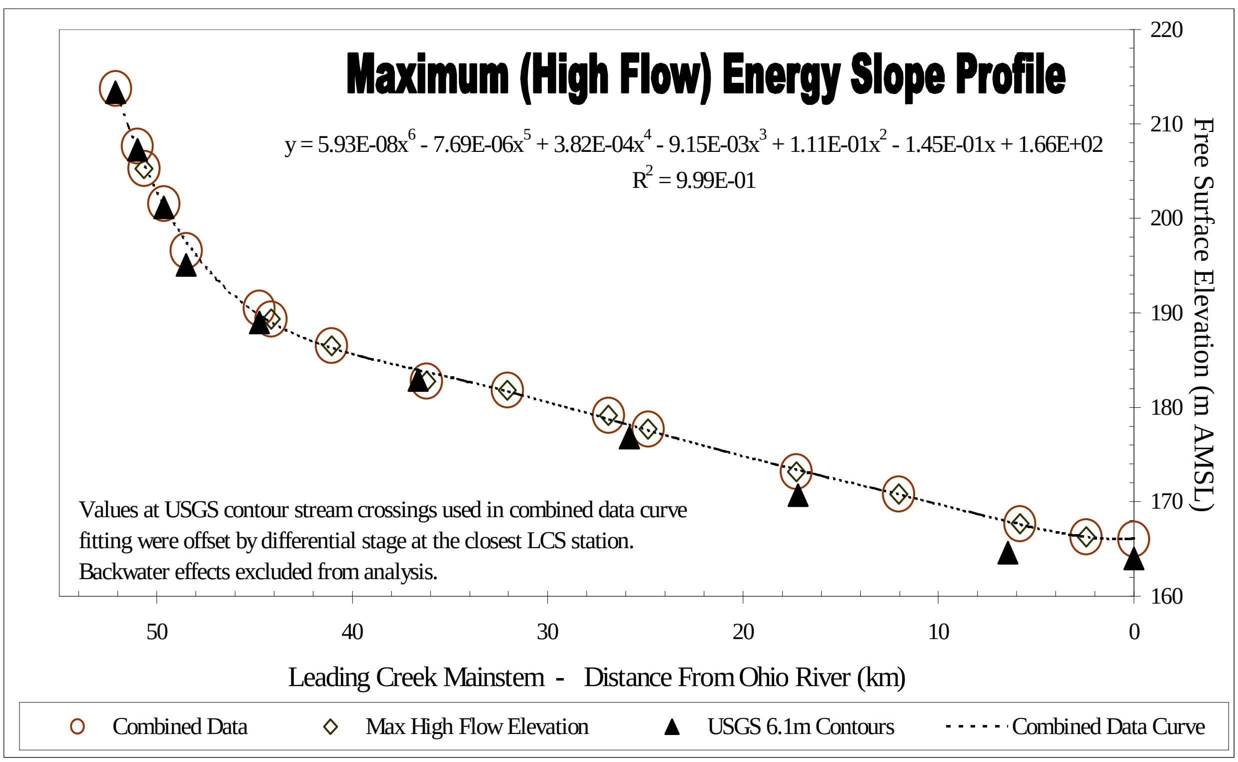


Figure F14 – Leading Creek Maximum High Flow Condition Elevation Model

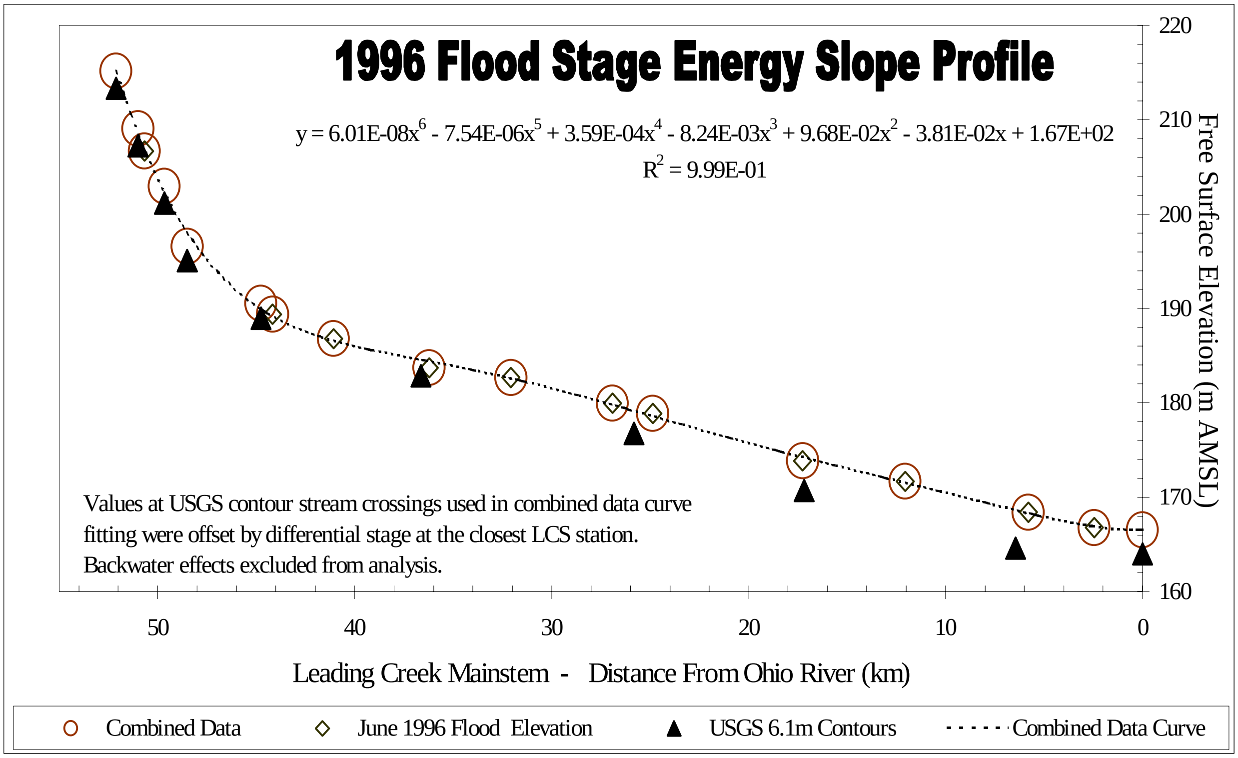


Figure F15 – Leading Creek June 1996 Flood Crest Elevation Model

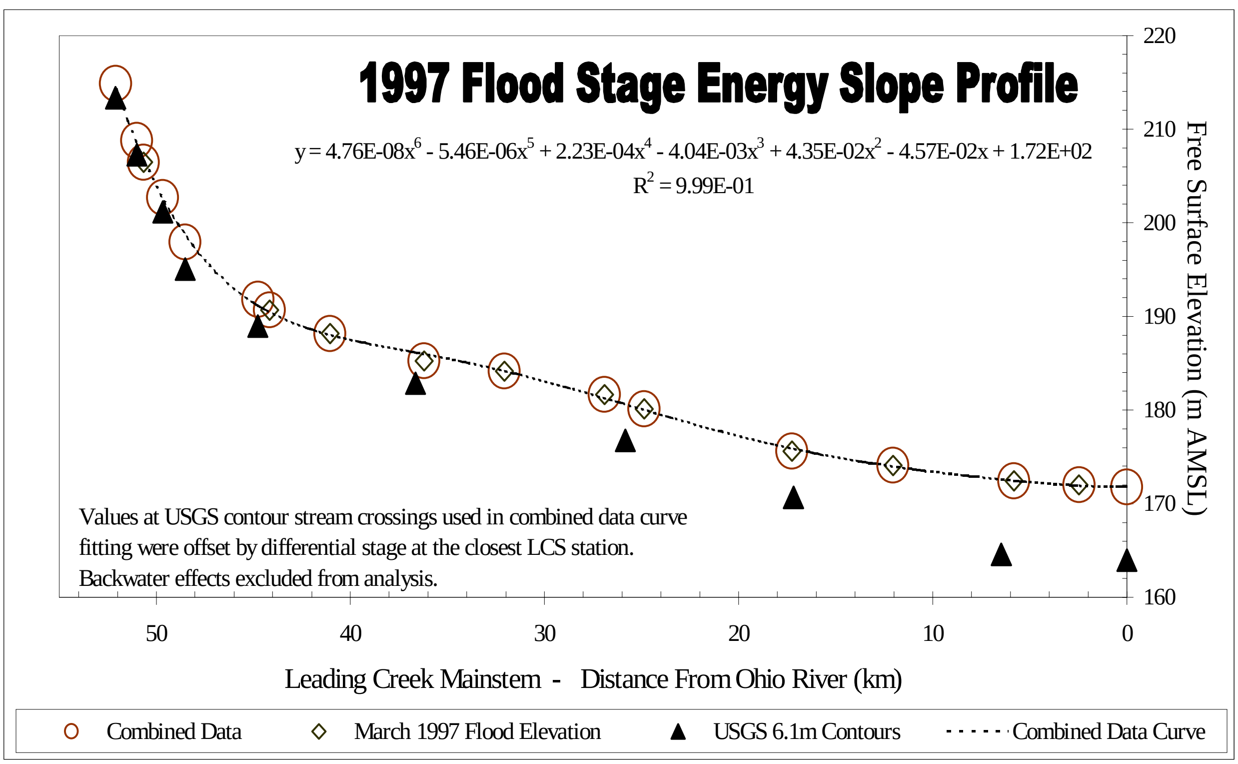


Figure F16 – Leading Creek March 1997 Flood Crest Elevation Model

Appendix G - Stage-Discharge Rating Curves

Justin Eric Babendreier

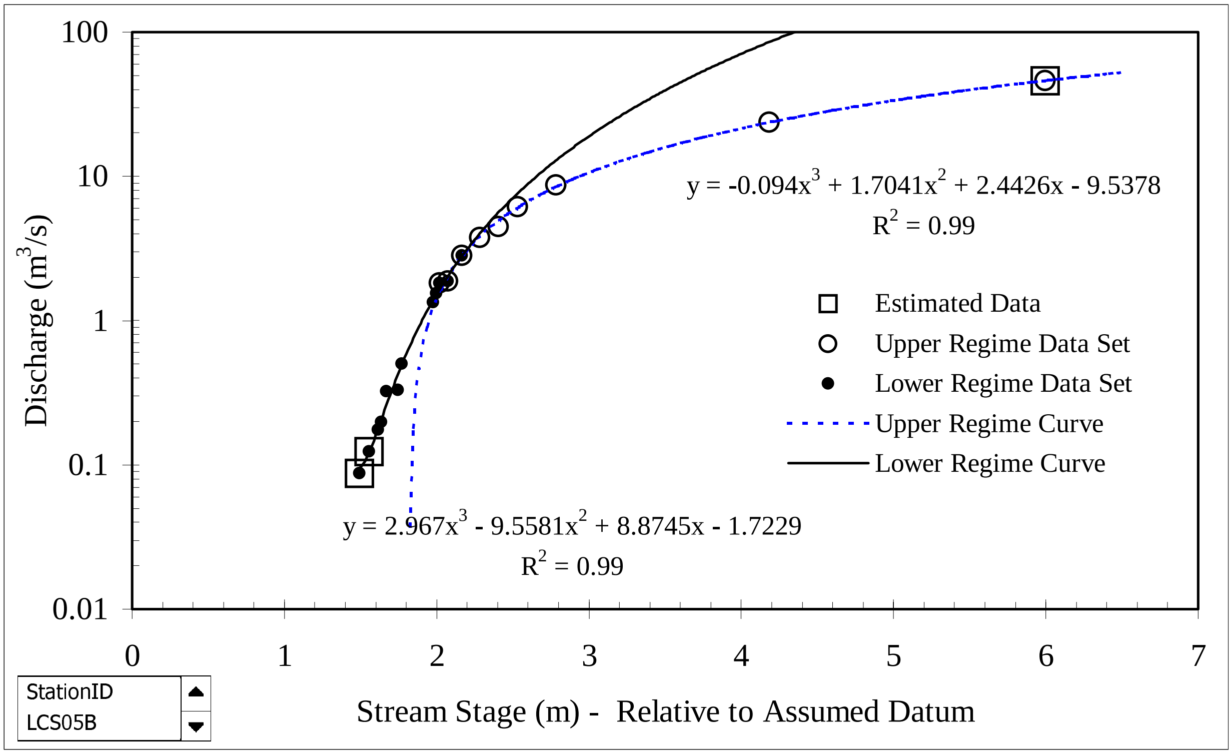


Figure G1 – Rating Curve for Leading Creek Continuous Gage Station LCS5B below Dexter, Ohio

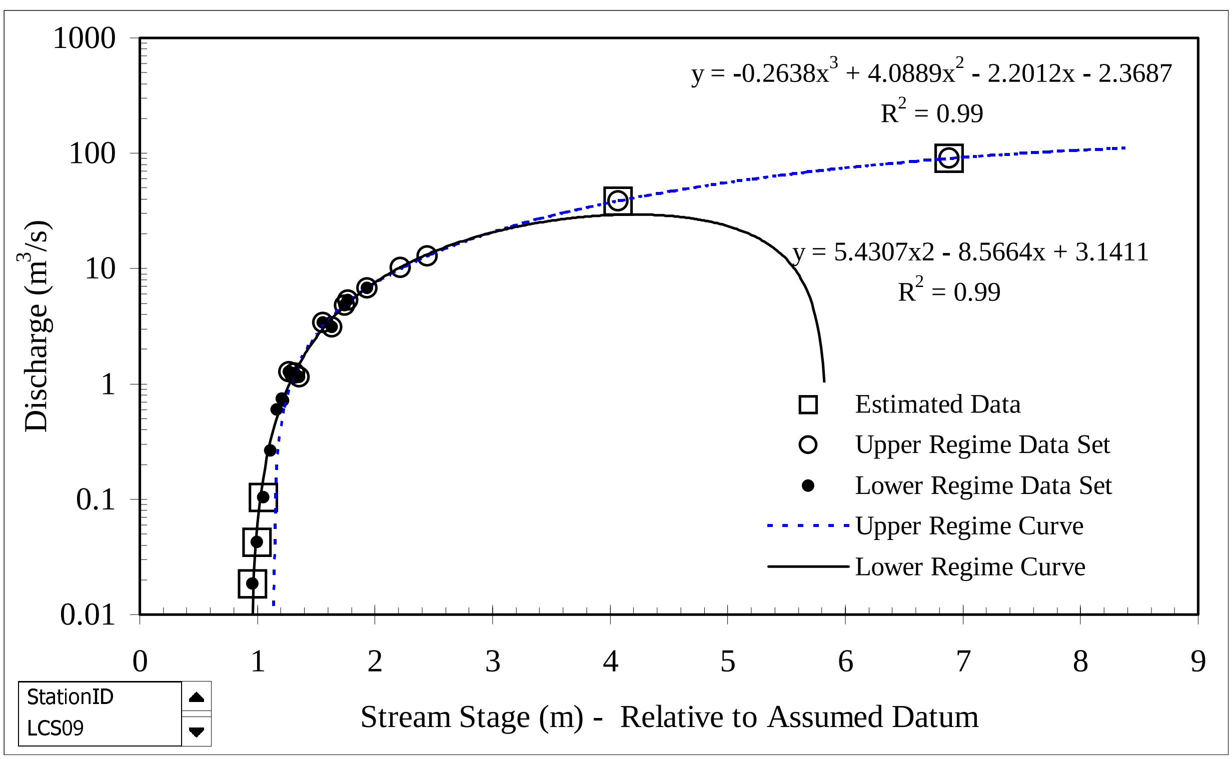


Figure G2 – Rating Curve for Leading Creek Continuous Gage Station LCS9 above Middleport, Ohio

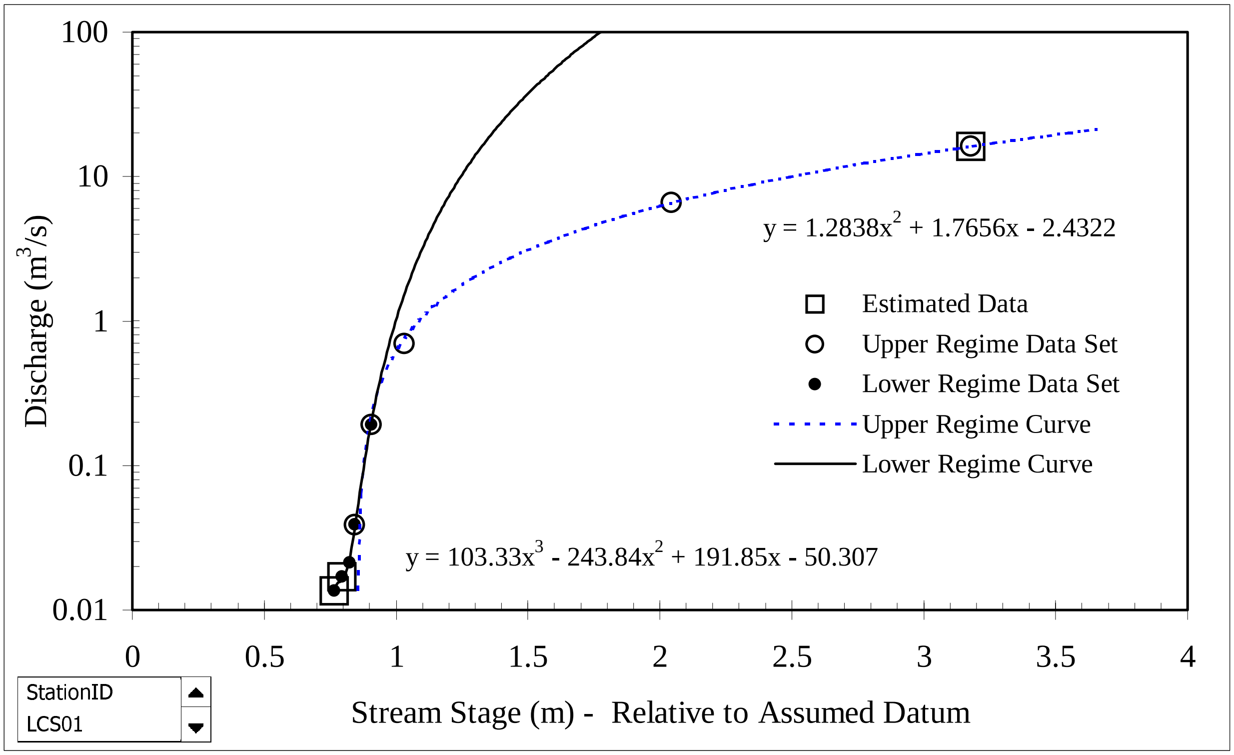


Figure G3 – Rating Curve for Leading Creek Manual Staff Gage Station LCS1 below Albany, Ohio

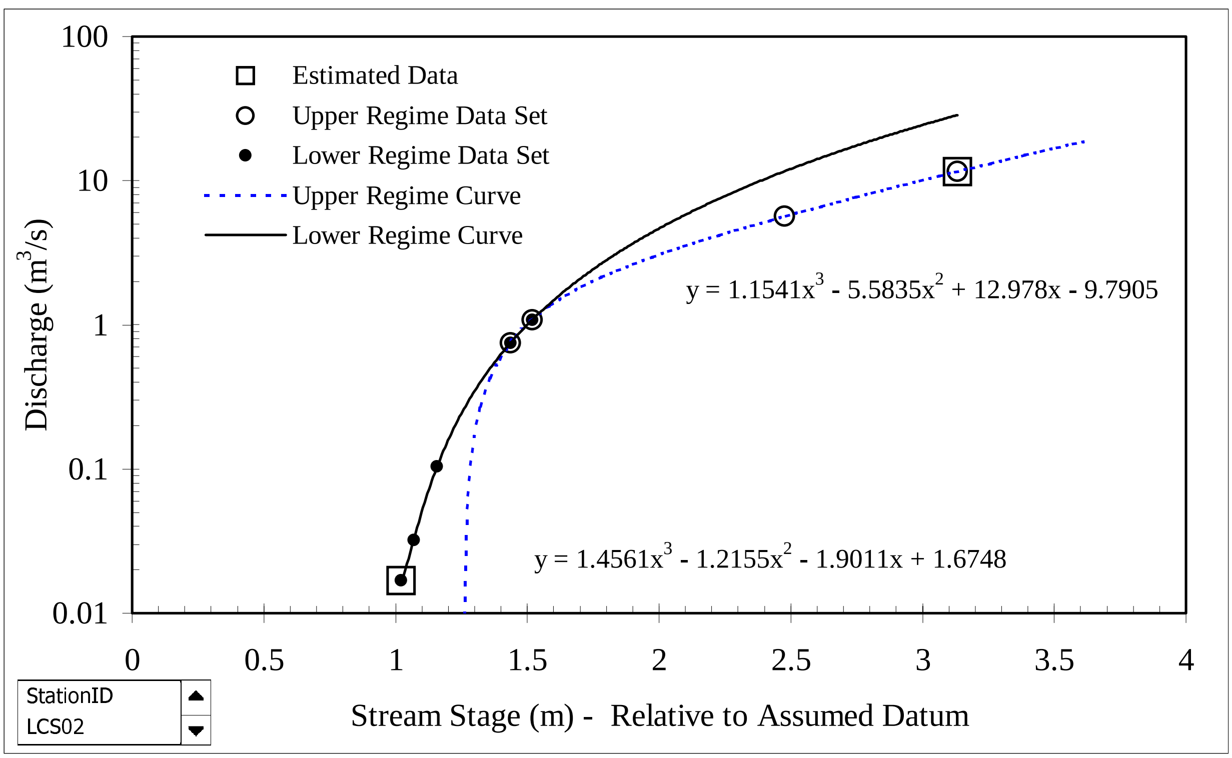


Figure G4 – Rating Curve for Leading Creek Manual Staff Gage Station LCS2 at Carpenter, Ohio

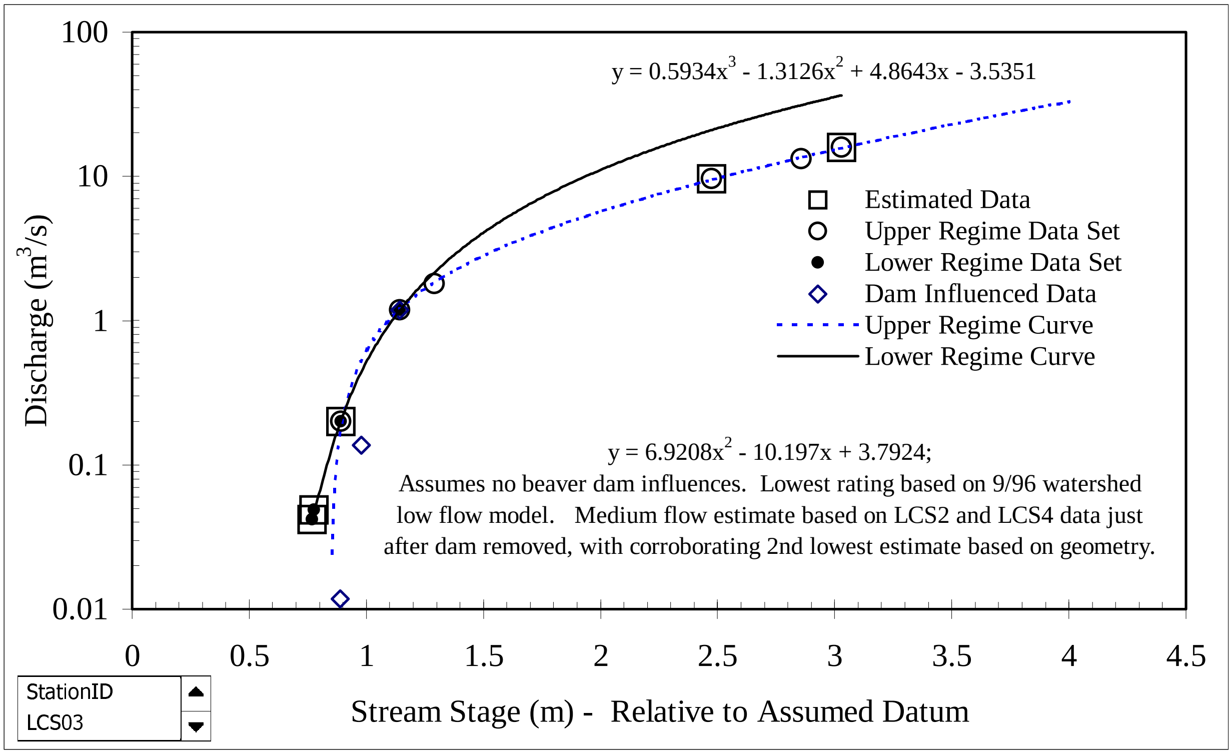


Figure G5 – Rating Curve for Leading Creek Manual Staff Gage Station LCS3 above Sisson Run below Carpenter, Ohio

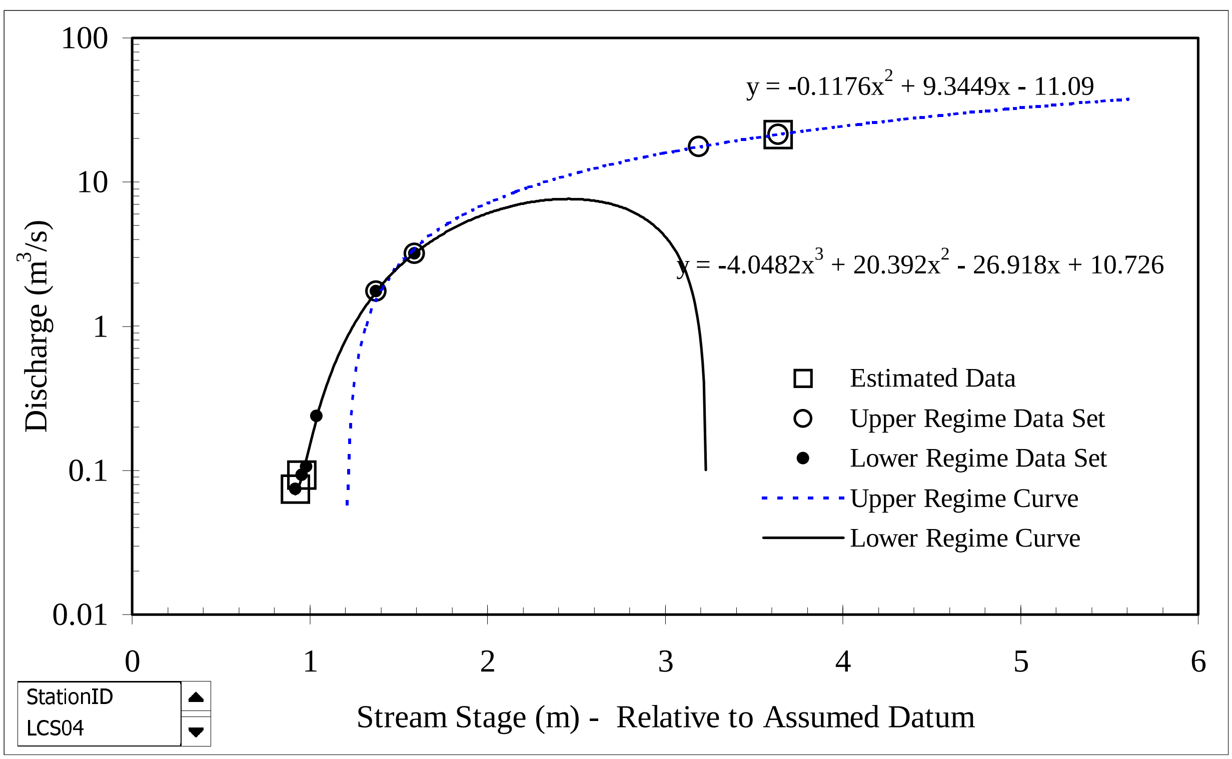


Figure G6 – Rating Curve for Leading Creek Manual Staff Gage Station LCS4 above Dyesville, Ohio

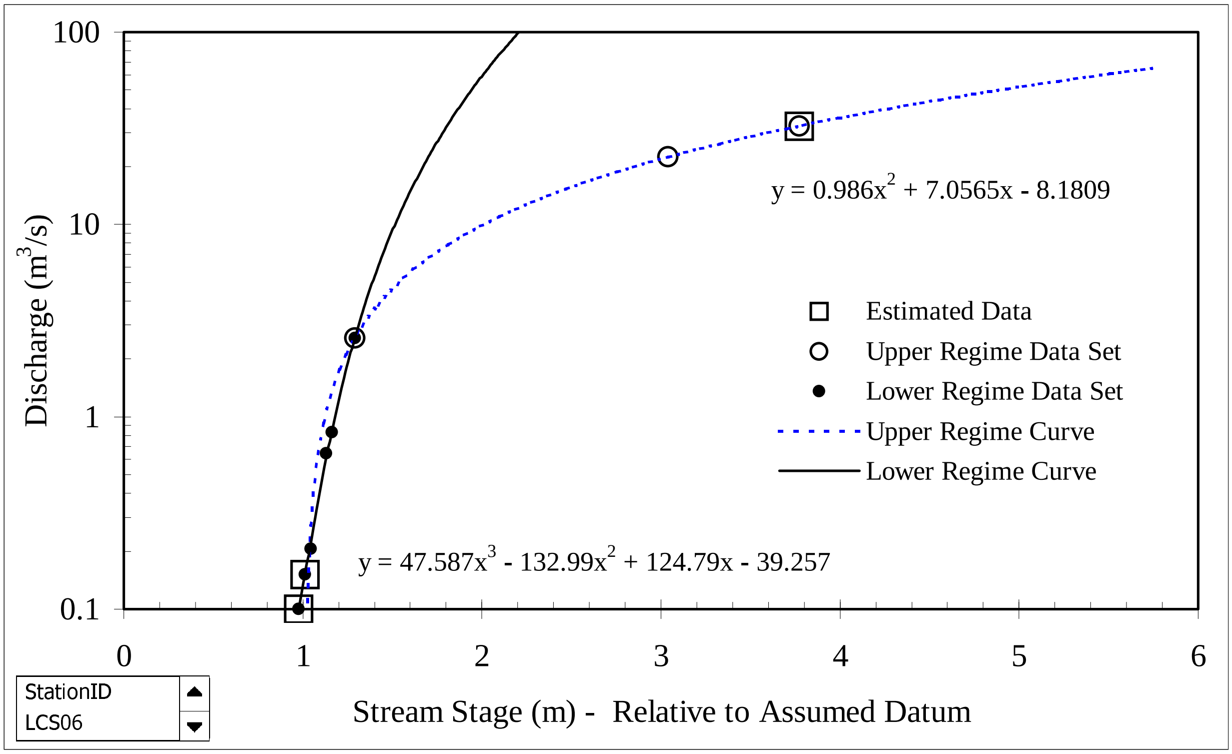


Figure G7 – Rating Curve for Leading Creek Manual Staff Gage Station LCS6 above Langesville, Ohio

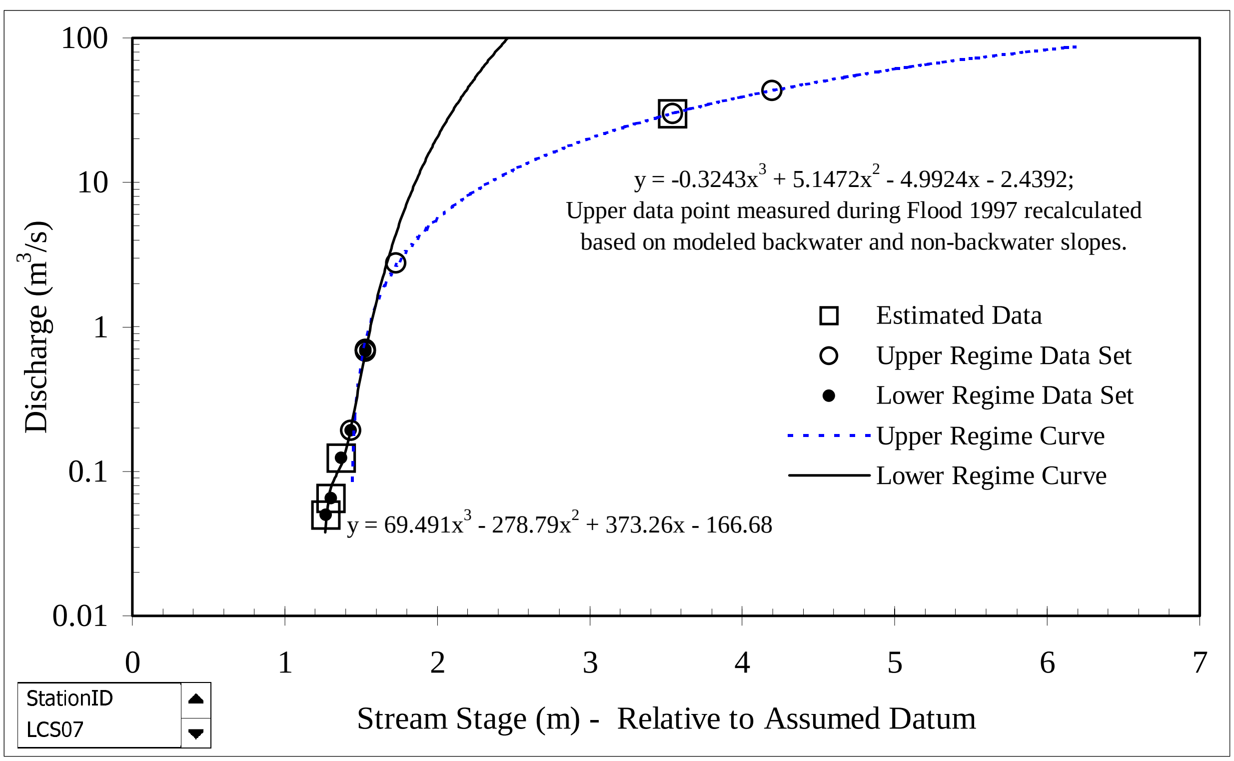


Figure G8 – Rating Curve for Leading Creek Manual Staff Gage Station LCS7 above Rutland, Ohio

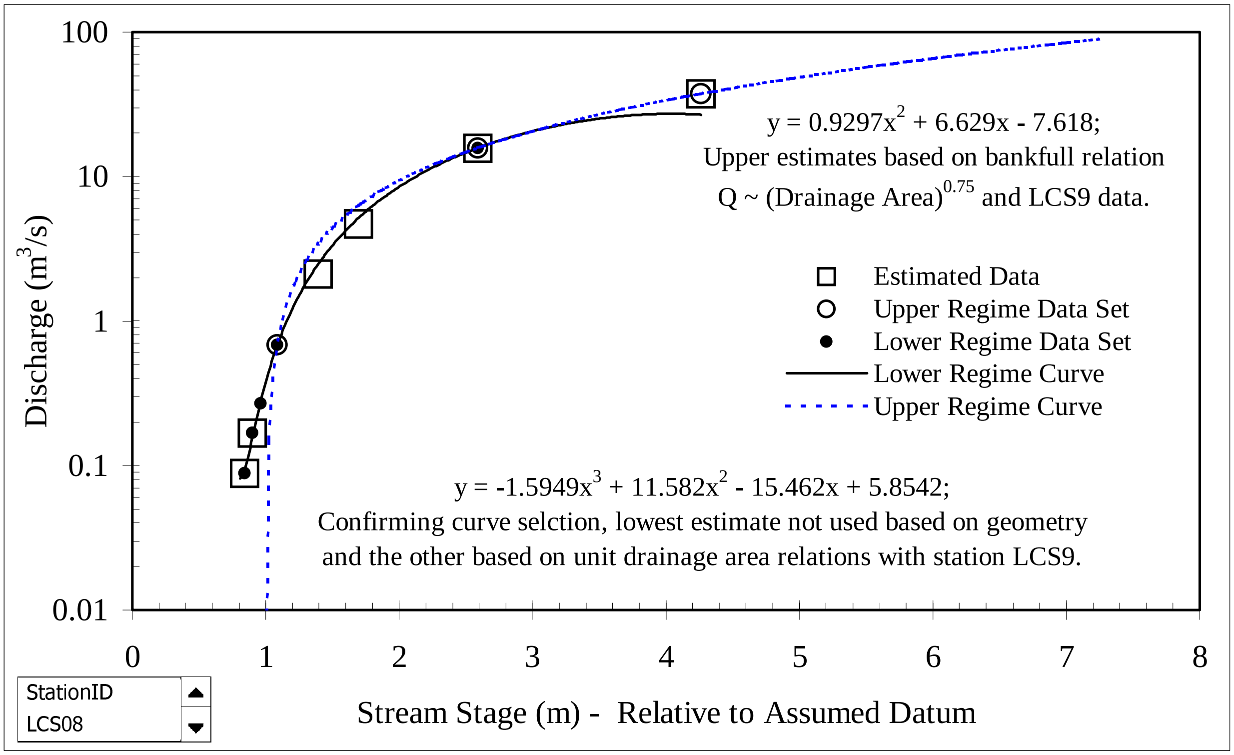


Figure G9 – Rating Curve for Leading Creek Manual Staff Gage Station LCS8 below Rutland, Ohio

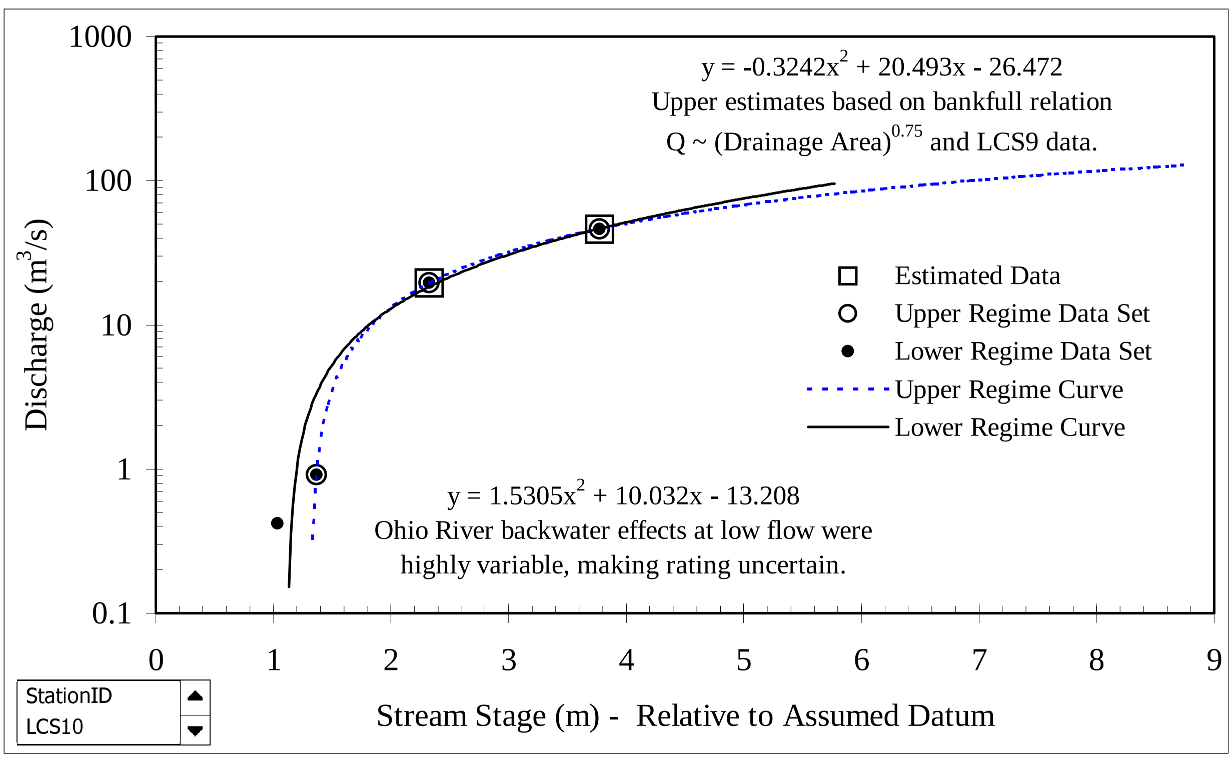


Figure G10 – Rating Curve for Leading Creek Manual Staff Gage Station LCS10 at Middleport, Ohio

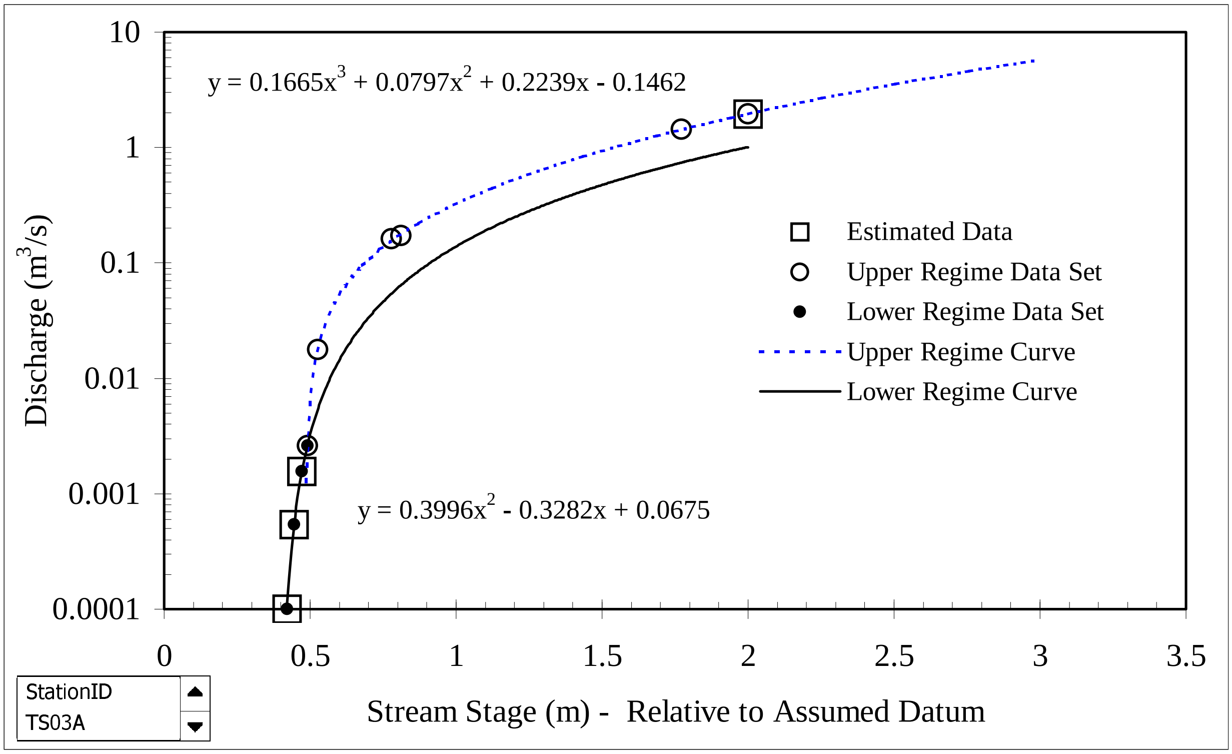


Figure G11 – Rating Curve for Manual Staff Gage Station TS3A on Five Mile Run near Carpenter, Ohio

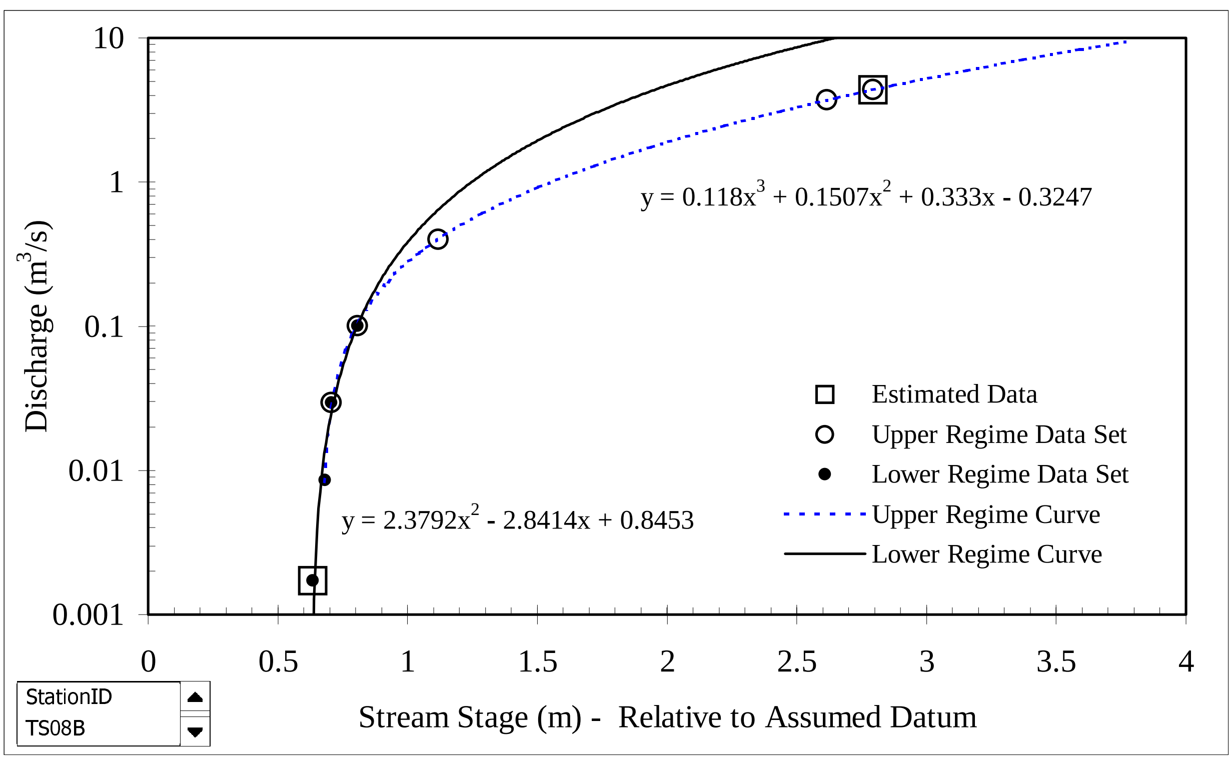


Figure G12 – Rating Curve for Manual Staff Gage Station TS8B on Mud Fork above Dexter, Ohio

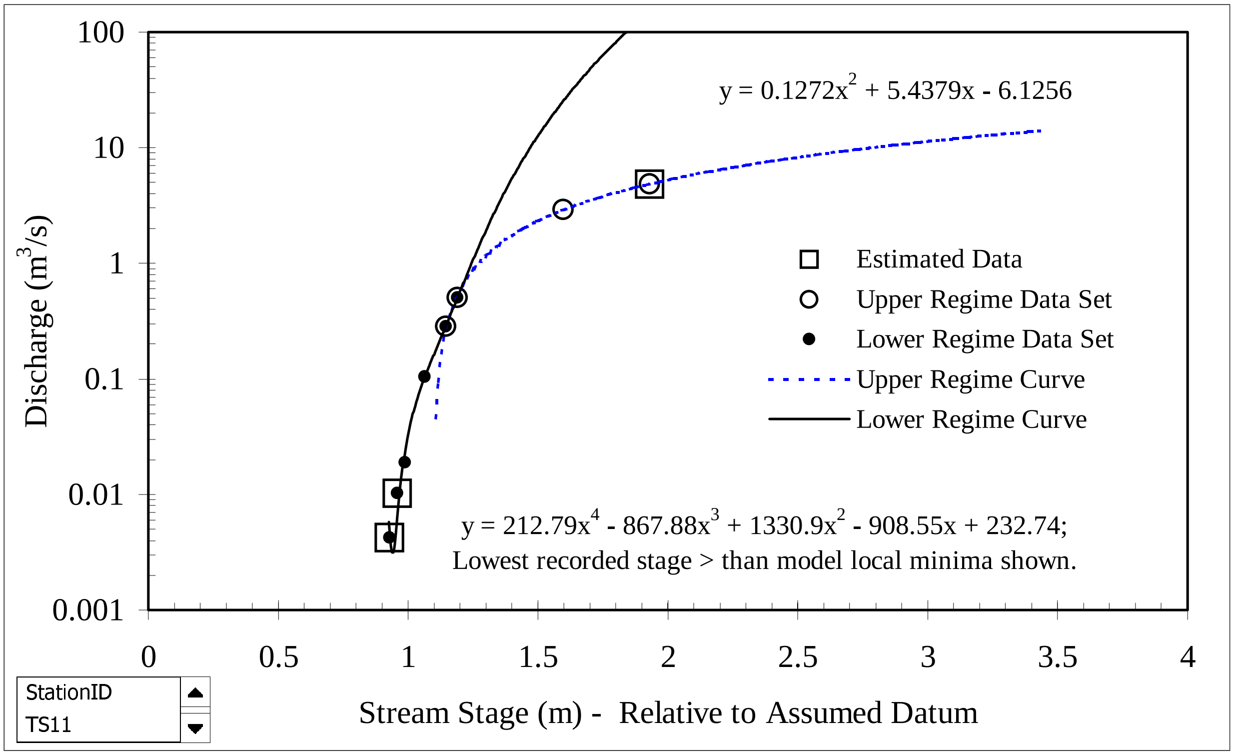


Figure G13 – Rating Curve for Manual Staff Gage Station TS11 on Parker Run near Dexter, Ohio

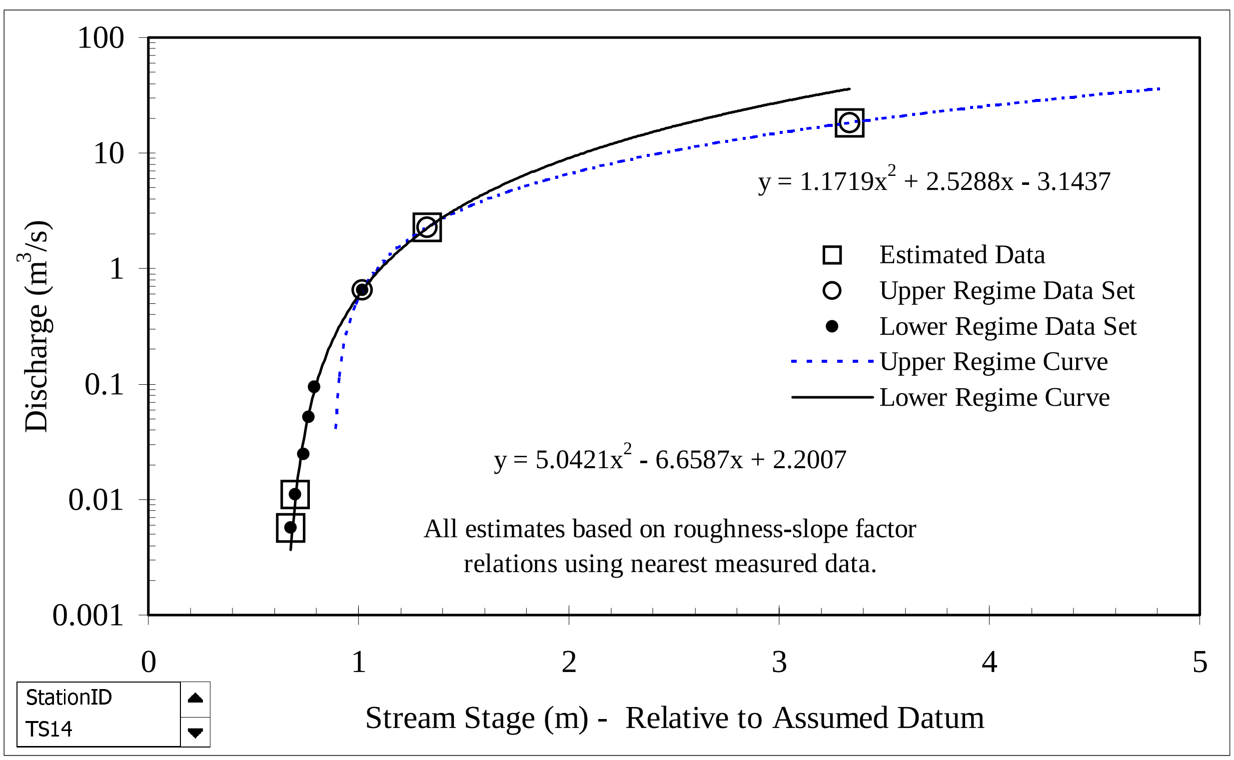


Figure G14 – Rating Curve for Manual Staff Gage Station TS14 on Little Leading Creek below Rutland, Ohio

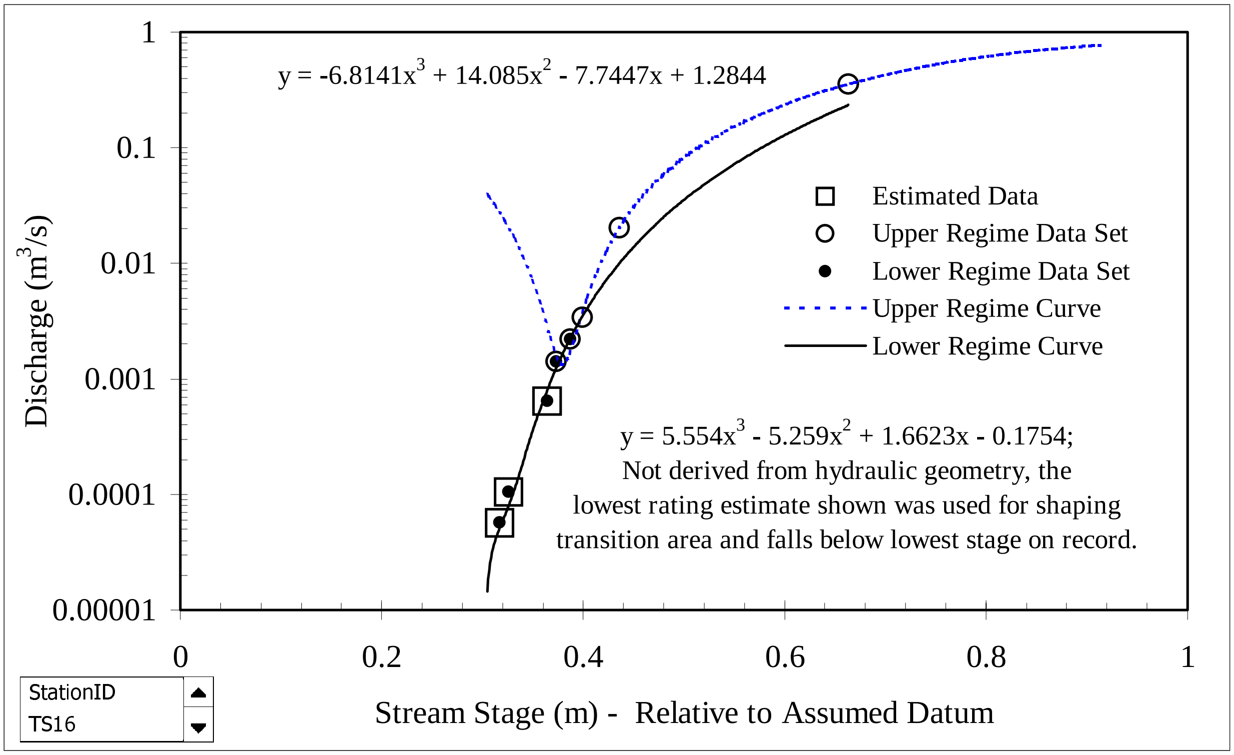


Figure G15 – Rating Curve for Manual Staff Gage Station TS16 on Paulins Hill Run near Middleport, Ohio

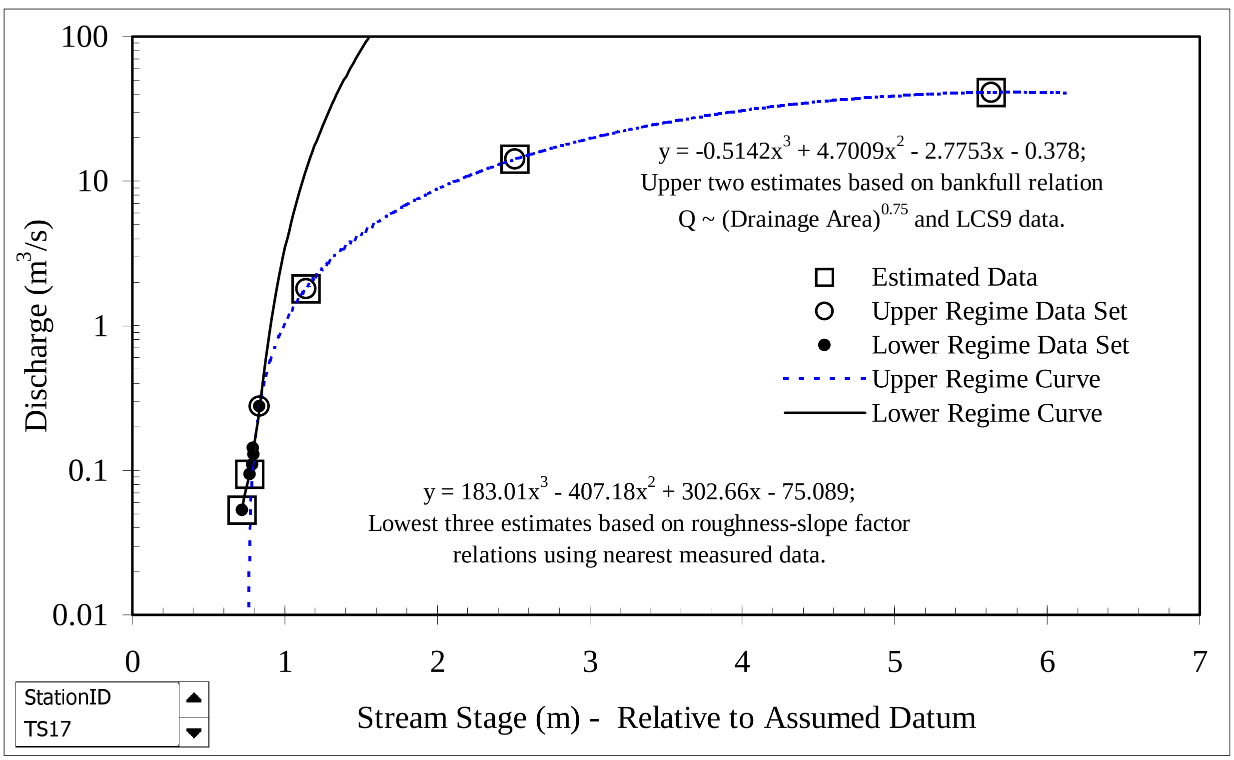


Figure G16 – Rating Curve for Manual Staff Gage Station TS17 on Thomas Fork near Middleport, Ohio

Table G1 – Lower-Upper Regime Rating Curve Match Point Summary. Station rating curves were based on best fit 2^0 to 4^0 polynomials used to characterize lower and upper regime discharge rates for the period of record 1996 to 1998. A limitation of the modeling technique, non-zero minima shown represent local curve minima. Application of models below actual rating data shown in accompanying stage-discharge figures should be viewed with added caution.

Staff Gage Station	Curve Match Point (m)	Lower Regime Minima (m)	Model Minima Flow (m^3/s)
LCS1	0.93	0.74	0.000
LCS2	1.52	0.99	0.014
LCS3	1.15	0.74	0.037
LCS4	1.46	0.90	0.066
LCS5B	2.11	1.47	0.093
LCS6	1.29	0.88	0.000
LCS7	1.47	1.25	0.000
LCS8	2.59	0.80	0.080
LCS9	1.86	0.95	0.000
LCS10	1.32	1.32	0.000
TS3A	0.49	0.41	0.000
TS8B	0.70	0.63	0.000
TS11	1.21	0.94	0.003
TS14	1.35	0.66	0.002
TS16	0.40	0.30	0.000
TS17	0.84	0.69	0.000

Appendix H - Sediment Chemistry and Toxicological Data

Justin Eric Babendreier

Table H1 – Average Sediment Quality Data for 1996 to early 1998. Standard deviations are given in parenthesis. Data presents site-specific conditions during annual sediment quality monitoring. Federal Creek was sampled three times in 1996 indicated by stations (R)FCS1, (R)FCS2, and (R)FCS3. Station TS7 was not sampled for some biological testing during the study due to access problems at the bridge site and confrontations with an adjacent landowner. Only 4 samples were analyzed for copper at TS6 in 1996.

Station ID	ASML	AUML	No.2	No.31	AG	Urban	# Cross-Section Samples		% Carbon				Copper (mg/kg)				Zinc (mg/kg)				Manganese (g/kg)				Iron (g/kg)			
							1996	1997	1996		1997		1996		1997		1996		1997		1996		1997		1996		1997	
(R)FCS1	x	x			x		5	0	0.41%	(0.14%)	-	-	30	(6.5)	-	-	75	(9.0)	-	-	1.1	(0.17)	-	-	43	(5.2)	-	-
(R)FCS2	x	x			x		5	0	0.37%	(0.17%)	-	-	55	(30)	-	-	83	(14)	-	-	1.0	(0.17)	-	-	49	(14)	-	-
(R)FCS3	x	x			x		5	0	-	-	-	-	21	(5.3)	-	-	74	(6.2)	-	-	1.1	(0.10)	-	-	44	(3.5)	-	-
LCS1					x	x	5	5	1.6%	(0.55%)	1.5%	(0.07%)	35	(14)	35	(5.1)	105	(13)	90	(4.1)	1.5	(0.21)	1.4	(0.077)	44	(3.7)	42	(1.3)
LCS2					x		5	5	0.25%	(0.05%)	1.1%	(0.31%)	15	(14)	16	(11)	22	(2.2)	39	(5.4)	0.32	(0.056)	1.0	(0.64)	10	(1.0)	18	(2.6)
LCS3					x		5	5	0.46%	(0.11%)	0.48%	(0.26%)	52	(32)	15	(7.5)	25	(2.3)	22	(6.5)	0.40	(0.043)	0.48	(0.094)	8.5	(0.31)	11	(2.9)
LCS4			x		x		5	5	0.15%	(0.06%)	0.20%	(0.03%)	18	(4.5)	12	(4.2)	23	(1.1)	31	(19)	0.23	(0.051)	0.47	(0.13)	8.8	(1.2)	13	(1.9)
LCS5B	x		x		x		5	5	0.26%	(0.21%)	0.34%	(0.02%)	25	(12)	14	(9.0)	51	(12)	36	(9.8)	0.74	(0.26)	0.65	(0.15)	26	(7.0)	23	(7.5)
LCS6	x		x	x	x		5	5	0.17%	(0.06%)	0.34%	(0.05%)	31	(10)	2.8	(1.5)	39	(4.9)	34	(1.7)	0.48	(0.047)	0.35	(0.055)	20	(4.0)	12	(0.83)
LCS7	x		x	x	x		4	5	0.13%	(0.02%)	0.13%	(0.01%)	6.5	(1.7)	21	(10)	19	(4.5)	41	(6.2)	0.30	(0.081)	0.46	(0.026)	11	(4.0)	17	(1.0)
LCS8	x	x	x	x	x		5	5	0.29%	(0.06%)	0.51%	(0.07%)	12	(6.8)	8.6	(4.3)	42	(5.7)	54	(2.9)	0.59	(0.045)	0.53	(0.027)	14	(0.8)	17	(0.95)
LCS9	x	x	x	x	x		5	5	-	-	0.40%	(0.44%)	10	(3.6)	22	(9.7)	59	(7.7)	71	(7.0)	1.0	(0.40)	1.3	(0.19)	15	(3.2)	17	(2.3)
LCS10	x	x	x	x	x		5	5	0.51%	(0.25%)	0.81%	(0.23%)	14	(3.2)	17	(10)	54	(14)	73	(13)	0.38	(0.10)	0.57	(0.32)	19	(5.3)	28	(13)
TS1					x	x	5	5	0.64%	(0.31%)	1.4%	(0.36%)	15	(1.6)	18	(4.7)	49	(8.6)	59	(6.5)	0.59	(0.21)	0.82	(0.11)	25	(3.4)	29	(2.5)
TS2					x	x	5	5	1.3%	(0.24%)	1.8%	(0.10%)	21	(5.7)	20	(5.4)	75	(6.6)	81	(4.3)	1.2	(0.25)	1.0	(0.34)	44	(4.7)	39	(4.6)
TS3B					x		5	5	0.28%	(0.18%)	0.06%	(0.01%)	17	(3.0)	11	(16)	38	(5.5)	22	(8.6)	0.77	(0.17)	0.26	(0.026)	20	(2.3)	7.8	(0.6)
TS4					x		5	5	0.12%	(0.03%)	0.68%	(0.54%)	14	(16)	18	(9.4)	25	(3.1)	25	(3)	0.18	(0.024)	0.28	(0.084)	8.1	(0.70)	12	(3.7)
TS5					x		5	5	0.06%	(0.01%)	0.07%	(0.05%)	20	(5.7)	2.2	(2.7)	17	(0.8)	15	(2.5)	0.27	(0.021)	0.24	(0.023)	8.1	(0.79)	7.2	(1.9)
TS6			x		x	x	5	5	0.10%	(0.03%)	0.08%	(0.01%)	18	(25)	3.2	(3.5)	28	(5.2)	33	(2.3)	0.37	(0.048)	0.45	(0.048)	16	(2.5)	21	(1.5)
TS8A	x				x		3	3	0.24%	(0.16%)	0.11%	(0.02%)	27	(12)	29	(3.5)	38	(3.0)	43	(12)	0.85	(0.19)	1.1	(0.27)	16	(2.4)	20	(6.7)
TS8B	x				x		3	5	0.25%	(0.05%)	0.26%	(0.05%)	26	(10)	2.2	(1.6)	52	(6.8)	48	(1.9)	0.84	(0.17)	1.0	(0.060)	16	(2.9)	15	(0.8)
TS9					x		5	5	1.1%	(0.73%)	0.93%	(0.06%)	30	(12)	8.6	(5.4)	60	(7.2)	50	(2.3)	0.53	(0.12)	0.35	(0.012)	28	(6.7)	21	(1.5)
TS10	x				x		5	3	0.26%	(0.12%)	1.5%	(0.28%)	43	(18)	32	(9.0)	78	(8.6)	95	(18)	1.2	(0.11)	1.7	(0.33)	27	(2.3)	28	(4.0)
TS11				x		x	5	5	0.46%	(0.11%)	0.96%	(0.10%)	62	(19)	30	(9.8)	143	(9.1)	119	(3.3)	2.0	(0.29)	2.1	(0.10)	86	(10)	75	(3.5)
TS12					x		5	3	0.56%	(0.37%)	1.0%	(0.11%)	29	(1.9)	28	(5.3)	74	(11)	67	(4.6)	1.1	(0.26)	0.88	(0.076)	43	(10)	34	(2.4)
TS13	x				x		5	3	0.22%	(0.05%)	0.18%	(0.04%)	18	(13)	16	(4.0)	33	(3.3)	32	(6.9)	0.59	(0.067)	0.63	(0.015)	16	(1.8)	18	(4.6)
TS14	x	x			x	x	5	5	0.25%	(0.10%)	0.15%	(0.04%)	27	(16)	17	(9.0)	75	(13)	68	(13)	1.6	(0.38)	1.3	(0.37)	17	(3.1)	14	(3.7)
TS15	x	x			x		5	5	0.22%	(0.06%)	0.17%	(0.06%)	22	(25)	11	(7.0)	37	(7.3)	29	(3.6)	0.61	(0.21)	0.38	(0.067)	16	(2.1)	15	(2.6)
TS16	x	x			x		5	5	0.45%	(0.26%)	0.31%	(0.06%)	19	(6.0)	20	(4.5)	33	(6.9)	48	(4.3)	0.24	(0.049)	0.63	(0.070)	23	(3.4)	17	(1.7)
TS17	x	x			x		5	5	0.44%	(0.09%)	0.50%	(0.07%)	21	(6.3)	26	(2.9)	75	(6.9)	86	(3.9)	0.17	(0.012)	0.18	(0.005)	37	(3.3)	40	(2.1)

Table H2 – Average Sediment Toxicity Data for 1996 to 1997. Standard deviations are given in parenthesis. Midge growth per test vessel (i.e. per subsection sample) was calculated as the average weight of recovered organisms from which event cross-section distributions were determined. Not utilized here, biased cross-section averages originally reported were calculated as the sum of average weights per sample (e.g. per test vessel) divided by the summed weight of all organisms per site-event (Cherry *et al*, 1991).

Station ID	ASML	AUML	No.2	No.31	AG	Urban	# Cross-Section Samples		*Daphnia magna*								*Chironomus tentans*							
									Neonate Production				% Survivorship				Growth (mg)				% Survivorship			
							1996	1997	1996		1997		1996		1997		1996		1997		1996		1997	
(R)FCS1	x	x			x		5	5	86	(5.9)	78	(12)	96%	(8.9%)	84%	(17%)	1.8	(1.1)	6.1	(2.2)	46%	(5.5%)	78%	(22%)
(R)FCS2	x	x			x		5	0	60	(20)	-	-	92%	(18%)	-	-	1.3	(0.73)	-	-	64%	(40%)	-	-
(R)FCS3	x	x			x		5	0	13	(5.3)	-	-	24%	(17%)	-	-	3.6	(0.36)	-	-	78%	(16%)	-	-
(R)SCS1	x	x			x		0	5	-	-	58	(36)	-	-	84%	(22%)	1.5	(0.50)	4.1	(0.79)	38%	(8.4%)	56%	(19%)
(R)SCS2	x	x			x		0	5	-	-	17	(20)	-	-	84%	(36%)	4.1	(0.25)	2.4	(0.75)	82%	(11%)	94%	(5.5%)
LCS1					x	x	5	5	84	(9.2)	77	(35)	96%	(8.9%)	72%	(30%)	1.1	(1.1)	2.4	(2.6)	40%	(37%)	30%	(29%)
LCS2					x		5	5	135	(37)	154	(19)	96%	(8.9%)	88%	(11%)	2.0	(0.53)	3.8	(0.67)	80%	(24%)	54%	(21%)
LCS3					x		5	5	82	(23)	147	(21)	92%	(18%)	88%	(11%)	1.5	(0.50)	2.3	(1.7)	84%	(17%)	28%	(24%)
LCS4			x		x		5	5	40	(8.3)	135	(28)	88%	(11%)	92%	(11%)	2.4	(0.43)	3.7	(3.1)	70%	(16%)	44%	(18%)
LCS5B	x		x		x		5	5	35	(28)	164	(30)	96%	(8.9%)	84%	(17%)	2.1	(0.69)	2.3	(2.2)	56%	(18%)	46%	(42%)
LCS6	x		x	x	x		5	5	56	(17)	26	(16)	96%	(8.9%)	60%	(42%)	1.7	(0.31)	5.6	(4.0)	72%	(19%)	30%	(24%)
LCS7	x		x	x	x		5	5	17	(25)	40	(26)	36%	(30%)	68%	(30%)	2.9	(1.7)	4.1	(0.79)	66%	(38%)	60%	(19%)
LCS8	x	x	x	x	x		5	5	45	(23)	76	(21)	72%	(18%)	100%	(0%)	4.1	(0.63)	6.1	(6.4)	94%	(8.9%)	52%	(30%)
LCS9	x	x	x	x	x		5	5	48	(9.2)	17	(15)	88%	(18%)	28%	(11%)	3.5	(0.74)	4.4	(1.9)	86%	(11%)	34%	(8.9%)
LCS10	x	x	x	x	x		5	5	41	(8.9)	55	(20)	80%	(20%)	84%	(36%)	4.5	(0.82)	1.9	(0.88)	90%	(10%)	68%	(15%)
TS1					x	x	5	5	50	(19)	46	(31)	96%	(8.9%)	100%	(0%)	2.3	(0.45)	2.6	(2.4)	92%	(11%)	38%	(44%)
TS2					x	x	5	5	51	(16)	26	(22)	96%	(8.9%)	96%	(8.9%)	1.7	(0.61)	2.9	(0.69)	84%	(26%)	64%	(30%)
TS3B					x		5	5	92	(24)	38	(12)	92%	(11%)	92%	(11%)	1.8	(0.89)	2.7	(1.6)	72%	(33%)	70%	(45%)
TS4					x		5	5	96	(24)	18	(17)	96%	(8.9%)	60%	(37%)	1.8	(0.26)	9.3	(12)	84%	(17%)	76%	(37%)
TS5					x		5	5	92	(13)	13	(18)	80%	(20%)	44%	(46%)	1.3	(0.60)	1.7	(1.2)	80%	(24%)	48%	(38%)
TS6			x		x	x	5	5	105	(21)	29	(18)	96%	(8.9%)	96%	(8.9%)	2.0	(0.22)	3.3	(0.45)	88%	(11%)	88%	(4.5%)
TS8A	x				x		5	5	46	(34)	2	(3.5)	76%	(43%)	64%	(41%)	1.1	(0.19)	3.9	(1.3)	74%	(13%)	70%	(22%)
TS8B	x				x		5	5	43	(3.4)	15	(17)	88%	(11%)	76%	(43%)	1.8	(0.34)	2.7	(0.76)	56%	(22%)	60%	(19%)
TS9					x		5	5	41	(25)	17	(8.3)	92%	(18%)	92%	(11%)	3.0	(0.85)	3.3	(1.9)	66%	(11%)	26%	(23%)
TS10	x				x		5	5	39	(18)	9	(9.5)	92%	(11%)	64%	(17%)	2.6	(0.69)	4.3	(1.8)	58%	(18%)	52%	(28%)
TS11				x		x	5	5	18	(16)	40	(16)	40%	(37%)	84%	(8.9%)	2.1	(0.41)	3.3	(0.65)	86%	(11%)	80%	(20%)
TS12					x		5	5	23	(16)	10	(9.3)	60%	(35%)	56%	(26%)	1.9	(0.23)	5.5	(2.6)	76%	(23%)	44%	(24%)
TS13	x				x		5	5	79	(15)	3	(4.2)	92%	(11%)	20%	(28%)	3.2	(0.82)	3.4	(0.93)	76%	(21%)	34%	(25%)
TS14	x	x			x	x	5	5	87	(30)	39	(20)	96%	(8.9%)	56%	(22%)	3.7	(0.89)	3.7	(1.9)	96%	(8.9%)	64%	(15%)
TS15	x	x			x		5	5	51	(13)	70	(43)	76%	(17%)	68%	(27%)	5.0	(1.3)	5.3	(1.9)	88%	(13%)	50%	(24%)
TS16	x	x			x		5	5	43	(28)	71	(39)	80%	(20%)	64%	(22%)	3.9	(0.58)	5.7	(7.6)	92%	(4.5%)	58%	(33%)
TS17	x	x			x		5	5	43	(28)	103	(28)	64%	(30%)	76%	(26%)	2.5	(0.59)	4.5	(1.8)	82%	(13%)	40%	(19%)

Table H3a – Inter-Test Regression Results for 1996 Sediment Toxicity; $\alpha = 0.05$.

Sediment Toxicity Test	*D. magna* Neonate Production			*D. magna* %Survivorship			*C. tentans* Growth		
	R^2	[P]	1-β	R^2	[P]	1-β	R^2	[P]	1-β
D. magna %Survivorship	0.41	1.4×10^{-4}	0.99	-	-	-	-	-	-
C. tentans Growth	0.10	0.087	0.40	0.08	0.14	0.31	-	-	-
C. tentans %Survivorship	<0.01	0.89	0.05	0.02	0.41	0.13	0.26	0.003	0.88

Table H3b – Inter-Test Regression Results for 1997 Sediment Toxicity; $\alpha = 0.05$.

Sediment Toxicity Test	*D. magna* Neonate Production			*D. magna* %Survivorship			*C. tentans* Growth		
	R^2	[P]	1-β	R^2	[P]	1-β	R^2	[P]	1-β
D. magna %Survivorship	0.19	0.017	0.69	-	-	-	-	-	-
C. tentans Growth	0.01	0.58	0.08	0.04	0.29	0.18	-	-	-
C. tentans %Survivorship	0.05	0.22	0.23	0.07	0.16	0.28	<0.01	0.60	0.08

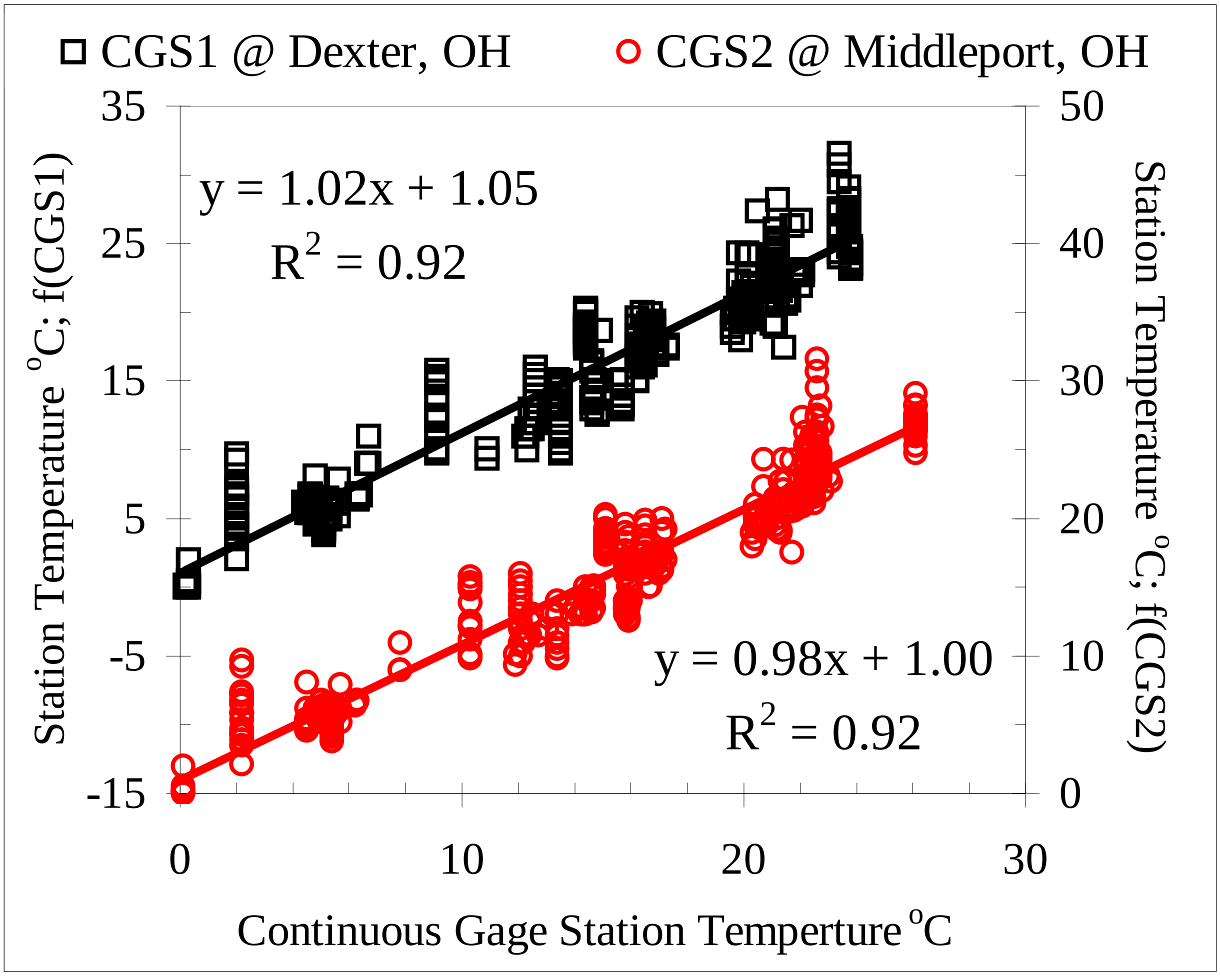


Figure H1 – Combined Station Water Temperature Model Results - Instantaneous CGS1 and CGS2 Data Matched to Routine Water Quality Monitoring - 1996 to 1997.

Table H4 – Individual Station Water Temperature Models – Routine Water Quality Monitoring - 1996 to 1997.

Station ID	CGS1/LCS5B (^oC)			CGS2/LCS9 (^oC)			n
	Slope	Intercept	R^2	Slope	Intercept	R^2	
AllRef Sites	1.07	0.00	0.95	1.01	0.27	0.95	12
(R)FCS1	1.00	1.95	0.62	0.93	2.42	0.64	4
(R)SCS1	1.01	0.00	1.00	0.96	0.38	1.00	4
(R)SCS2	1.05	-0.04	1.00	1.00	-0.10	0.99	4
LCS1	1.05	0.08	0.92	1.01	0.42	0.93	15
LCS2	1.06	-0.02	0.95	1.02	0.31	0.96	15
LCS3	1.08	-0.34	0.96	1.05	-0.16	0.96	14
LCS4	1.08	-0.29	0.96	1.04	-0.03	0.97	16
LCS5B	1.00	1.19	0.97	0.96	1.26	0.96	14
LCS6	1.03	1.29	0.97	0.98	1.43	0.95	14
LCS7	1.08	0.89	0.98	1.02	1.03	0.97	15
LCS8	1.05	0.78	0.98	1.00	0.80	0.97	16
LCS9	1.07	0.10	0.99	1.03	0.15	0.99	16
LCS10	1.04	0.12	0.98	1.00	0.10	0.98	16
TS1	1.02	0.23	0.91	0.99	0.21	0.92	15
TS2	1.07	-0.61	0.91	1.06	-0.72	0.93	15
TS3B	1.05	-0.63	0.93	1.01	-0.04	0.94	13
TS4	1.04	-0.09	0.95	1.01	0.13	0.97	14
TS5	0.97	0.60	0.90	0.97	0.49	0.91	12
TS6	1.09	-0.89	0.96	1.06	-0.62	0.97	12
TS7	0.99	0.00	0.96	0.98	0.00	0.96	11
TS8A	1.11	1.83	0.89	1.10	1.59	0.89	13
TS8B	1.09	1.19	0.93	1.04	1.17	0.92	13
TS9	1.09	0.31	0.93	1.04	0.37	0.93	13
TS10	0.98	1.92	0.94	0.92	2.13	0.93	13
TS11	1.11	2.43	0.94	1.08	2.17	0.93	13
TS12	0.99	1.51	0.96	0.94	1.56	0.95	14
TS13	1.07	2.09	0.92	1.02	2.20	0.91	15
TS14	1.03	0.70	0.97	0.99	0.63	0.96	16
TS15	0.92	2.74	0.91	0.90	2.56	0.91	13
TS16	1.00	1.50	0.95	0.95	1.54	0.95	14
TS17	1.00	0.76	0.96	0.95	0.77	0.97	15
Average	1.04	0.69	0.94	1.00	0.78	0.94	13
Std. Deviation	0.05	0.96	0.07	0.05	0.90	0.06	3.3

Table H5 – Clam Toxicity Testing Results - 1996 to 1997 – Site-event standard deviations of average shell length growth of 25 clams in 5 sample bags given in parenthesis; 35-day *in situ* placement. Station TS7 was not sampled for some biological testing during the study due to access problems at the bridge site and confrontations with an adjacent landowner. Data calculates average growth by assuming zero growth for dead clams. Dead clams are excluded from growth statistics however in cases where site mortality was below an expected natural population mortality of 20% for sites typically considered non-impacted.

Station ID	June 1996				August1996				June 1997				August 1997			
	Growth (mm)		%Surv.	Avg °C	Growth (mm)		%Surv.	Avg °C	Growth (mm)		%Surv.	Avg °C	Growth (mm)		%Surv.	Avg °C
(R)FCS1	0	(0)	0%	22.0	0	(0)	0%	23.1	0.1	(0.2)	36%	20.8	-- Not Sampled --			
(R)SCS1	-- Not Sampled --				-- Not Sampled --				0.6	(0.3)	92%	19.0	1.5	(0.6)	96%	21.1
LCS01	0.8	(0.4)	88%	21.1	0.5	(0.3)	76%	22.3	1.1	(0.4)	96%	19.9	0.9	(0.3)	100%	22.1
LCS02	0.7	(0.8)	48%	21.2	0.2	(0.4)	16%	22.4	0.9	(0.9)	96%	20.0	1.0	(0.7)	88%	22.2
LCS03	1.2	(0.7)	80%	21.3	0.3	(0.4)	36%	22.5	1.5	(0.5)	100%	20.0	1.3	(0.7)	100%	22.3
LCS04	2.0	(0.9)	84%	21.4	0.2	(0.6)	12%	22.6	0.2	(0.5)	20%	20.1	2.1	(0.7)	100%	22.3
LCS05B	0.7	(0.9)	40%	21.2	0.2	(0.5)	24%	22.4	0.1	(0.2)	30%	20.0	1.7	(0.3)	100%	22.1
LCS06	2.4	(1.1)	100%	21.9	1.7	(0.7)	88%	23.1	0.0	(0.0)	20%	20.7	0.7	(0.3)	78%	22.8
LCS07	2.5	(0.8)	100%	22.5	0.3	(0.6)	24%	23.7	0.9	(0.7)	84%	21.2	1.3	(0.7)	92%	23.5
LCS08	2.3	(1.3)	76%	21.8	1.2	(0.7)	76%	23.0	0.0	(0)	0%	20.6	1.9	(0.3)	100%	22.8
LCS09	3.1	(1.0)	100%	21.5	1.8	(1.0)	76%	22.7	0.2	(0.1)	96%	20.3	1.8	(0.5)	100%	22.5
LCS10	0	(0)	0	21.0	0.6	(1.0)	28%	22.1	-- Lost --				1.9	(0.2)	100%	21.9
TS01	0.2	(0.4)	44%	20.7	0	(0)	0%	21.8	0.5	(0.6)	72%	19.5	0.2	(0.2)	68%	21.6
TS02	0.2	(0.3)	52%	20.8	0.4	(0.4)	60%	22.0	1.4	(0.9)	100%	19.6	0.4	(0.6)	33%	21.8
TS03B	0.2	(0.6)	16%	20.4	0.3	(0.6)	16%	21.6	1.3	(0.8)	100%	19.2	0.1	(0.2)	29%	21.3
TS04	0	(0)	0%	20.8	0.4	(0.6)	32%	21.9	0.3	(0.7)	20%	19.5	1.1	(0.6)	92%	21.7
TS05	0.2	(0.5)	20%	20.0	0.4	(0.6)	48%	21.1	-- Lost --				0.3	(0.4)	48%	20.9
TS06	1.4	(0.9)	84%	21.0	1.6	(0.4)	96%	22.2	0.6	(0.8)	84%	19.7	1.2	(0.9)	100%	21.9
TS07	0	(0)	0%	19.8	-- Not Sampled --				-- Not Sampled --				-- Not Sampled --			
TS08A	0.1	(0.2)	12%	24.1	-- Lost --				0.0	(0)	0%	22.8	-- Not Sampled - No Flow --			
TS08B	0	(0)	0%	23.0	0.3	(0.5)	48%	24.3	0.0	(0)	0%	21.7	1.6	(0.2)	100%	24.0
TS09	0.7	(0.5)	72%	22.2	0.6	(0.4)	80%	23.4	0.9	(0.4)	100%	20.9	1.5	(0.9)	100%	23.1
TS10	1.0	(0.3)	96%	21.6	1.2	(0.8)	64%	22.7	0.1	(0.3)	32%	20.4	1.0	(0.5)	100%	22.4
TS11	1.5	(0.8)	88%	24.7	2.1	(1.3)	72%	25.9	0.1	(0.2)	44%	23.4	0.2	(0.2)	48%	25.7
TS12	0.6	(0.6)	56%	21.4	0	(0)	0%	22.5	-- Lost --				1.1	(0.4)	100%	22.2
TS13	0.1	(0.1)	100%	23.5	0.1	(0.1)	96%	24.7	0.4	(0.5)	84%	22.3	0.3	(0.2)	79%	24.5
TS14	0.1	(0.2)	84%	21.3	0.8	(0.4)	92%	22.5	0.2	(0.3)	100%	20.1	1.5	(0.3)	100%	22.3
TS15	0	(0.0)	0%	21.2	0.1	(0.2)	92%	22.2	-- Lost --				-- Not Sampled - No Flow --			
TS16	0	(0.0)	0%	21.5	0	(0)	0%	22.7	-- Lost --				0	(0)	0%	22.4
TS17	0	(0.0)	0%	20.8	0	(0)	0%	21.9	0.0	(0)	0%	19.6	0	(0)	0%	21.7

Table H6 – 48-Hour Acute Toxicity Test Results for Storm Water Monitoring – 1997.

Station ID	Survivorship of *Ceriodaphnia dubia* at Dilution						Full Lethal Concentration Test 48 Hour LC50		
	10% Concentration			100% Concentration					
	Jun-97	Oct-97	Average	Jun-97	Oct-97	Average	Jun-97	Oct-97	Average
(R)SCS1	100%	-	100%	100%	100%	100%	-	-	-
LCS1	100%	100%	100%	100%	100%	100%	-	-	-
LCS2	100%	100%	100%	100%	100%	100%	-	-	-
LCS3	100%	100%	100%	100%	100%	100%	-	-	-
LCS4	95%	100%	98%	95%	100%	98%	-	-	-
LCS5B	100%	100%	100%	100%	100%	100%	-	-	-
LCS6	100%	100%	100%	100%	100%	100%	-	-	-
LCS7	60%	100%	80%	100%	100%	100%	-	-	-
LCS8	100%	100%	100%	100%	100%	100%	-	-	-
LCS9	100%	100%	100%	100%	100%	100%	-	-	-
LCS10	100%	90%	95%	100%	45%	73%	-	84%	84%
TS1	100%	60%	80%	100%	35%	68%	-	5.6%	5.6%
TS2	100%	100%	100%	100%	100%	100%	-	-	-
TS3B	100%	100%	100%	55%	100%	78%	None Gen.	-	-
TS4	100%	100%	100%	100%	100%	100%	-	-	-
TS5	100%	100%	100%	100%	100%	100%	-	-	-
TS6	100%	100%	100%	100%	55%	78%	-	None Gen.	-
TS8A	95%	Dry	95%	100%	Dry	100%	-	-	-
TS8B	100%	100%	100%	100%	100%	100%	-	-	-
TS9	100%	100%	100%	100%	95%	98%	-	-	-
TS10	100%	100%	100%	100%	95%	98%	-	-	-
TS11	100%	95%	98%	100%	90%	95%	-	-	-
TS12	100%	100%	100%	100%	100%	100%	-	-	-
TS13	100%	100%	100%	100%	100%	100%	-	-	-
TS14	65%	100%	83%	100%	100%	100%	-	-	-
TS15	100%	100%	100%	50%	100%	75%	58%	-	-
TS16	100%	-	100%	0%	-	0%	68%	6.9%	38%
TS17	60%	-	60%	0%	-	0%	37%	19%	28%

Appendix I - Water Chemistry Data

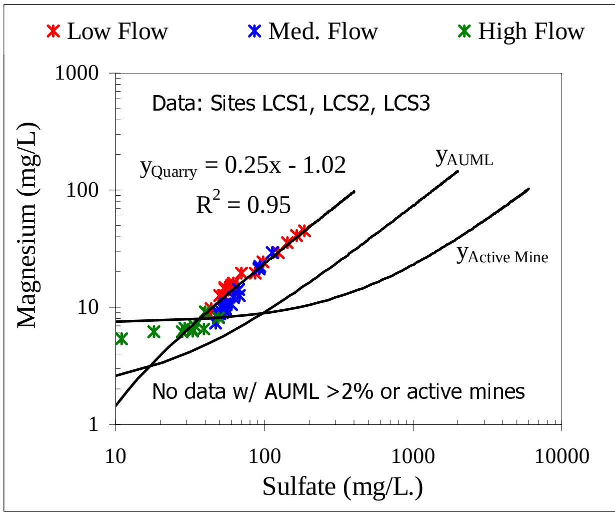
Justin Eric Babendreier

Table I1a – Average Monthly Water Quality Data for 1996 to 1997; All Flow Conditions. Standard deviations are given in parenthesis. For metals n = 15 to 16, for indicators n = 17 to 20; for reference sites n ≥ 5; Mg^{2+} and Ca^{2+} hardness reported as $CaCO_3$. Urban use defined by Ohio small town influences or sanitary NPDES discharge; Nos.2 and 31 are active underground long-wall mines below major drainage; AUML - abandoned underground mine lands above major drainage; ASML - abandoned surface contour mine land; and AG - agricultural activities. LCS1 and sites below to a significantly lessor degree are also influenced by a closed quarry operation.

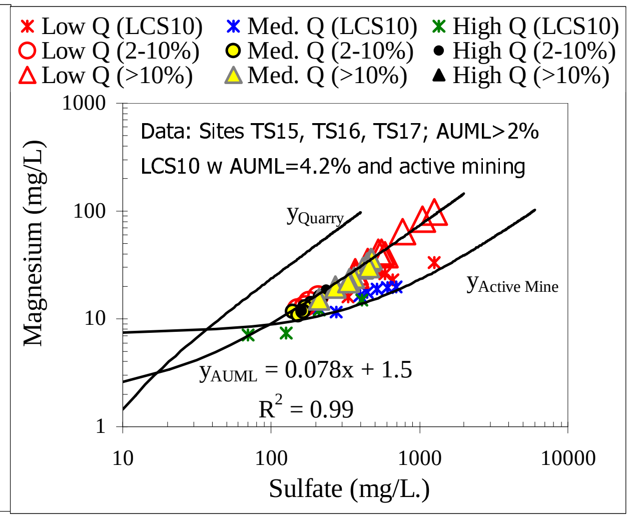
Station ID	ASML	AUML	No.2	No.31	AG	Urban	pH (s.u.)		Conductivity (mmho/cm)		Iron (mg/L)		Manganese (mg/L)		Zinc (ug/L)		Copper (ug/L)		Hardness (mg/L)	
(R)FCS1	x	x			x		7.8	(0.2)	0.44	(0.12)	3.25	(6.51)	0.19	(0.24)	16	(28)	3	(4)	207	(39)
(R)SCS1	x	x			x		7.3	(0.3)	0.30	(0.08)	0.83	(0.23)	0.37	(0.10)	7	(3)	2	(1)	111	(10)
(R)SCS2	x	x			x		7.3	(0.1)	0.23	(0.02)	1.51	(1.00)	0.19	(0.04)	8	(4)	2	(1)	109	(12)
LCS1					x	x	7.9	(0.4)	0.51	(0.25)	0.88	(1.31)	0.08	(0.07)	8	(12)	2	(1)	232	(102)
LCS2					x		7.7	(0.2)	0.40	(0.14)	1.49	(1.99)	0.28	(0.15)	9	(12)	2	(2)	163	(49)
LCS3					x		7.6	(0.4)	0.35	(0.11)	1.58	(2.29)	0.24	(0.12)	8	(9)	2	(1)	143	(34)
LCS4			x		x		7.6	(0.3)	0.42	(0.18)	1.57	(1.80)	0.29	(0.19)	8	(8)	2	(2)	144	(41)
LCS5B	x		x		x		7.7	(0.3)	0.43	(0.13)	1.09	(0.51)	0.33	(0.34)	6	(3)	1	(1)	140	(23)
LCS6	x		x	x	x		7.7	(0.3)	2.50	(1.78)	1.79	(3.83)	0.28	(0.14)	8	(8)	2	(3)	280	(116)
LCS7	x		x	x	x		7.6	(0.2)	2.25	(1.78)	1.86	(3.86)	0.26	(0.18)	8	(9)	4	(7)	265	(116)
LCS8	x	x	x	x	x		7.6	(0.2)	2.23	(1.63)	1.49	(2.76)	0.30	(0.13)	9	(9)	2	(2)	262	(105)
LCS9	x	x	x	x	x		7.5	(0.3)	2.11	(1.59)	1.09	(1.09)	0.28	(0.18)	10	(10)	2	(2)	264	(101)
LCS10	x	x	x	x	x		7.3	(0.3)	1.65	(1.38)	2.77	(4.74)	0.83	(0.54)	42	(25)	4	(4)	255	(85)
TS1					x	x	7.4	(0.3)	0.35	(0.20)	0.96	(0.71)	0.64	(0.77)	7	(6)	1	(1)	104	(25)
TS2					x	x	7.5	(0.3)	0.34	(0.20)	0.81	(0.68)	0.26	(0.34)	5	(4)	2	(1)	113	(27)
TS3B					x		7.6	(0.4)	0.35	(0.12)	1.82	(2.07)	1.08	(1.80)	9	(12)	2	(2)	139	(33)
TS4					x		7.5	(0.3)	0.24	(0.06)	1.34	(1.52)	0.35	(0.17)	7	(8)	1	(1)	99	(18)
TS5					x		7.6	(0.3)	0.22	(0.05)	1.18	(1.38)	0.22	(0.12)	7	(6)	2	(2)	91	(16)
TS6			x		x	x	7.6	(0.2)	1.02	(1.09)	1.28	(1.72)	0.19	(0.08)	7	(6)	2	(1)	165	(90)
TS7					x		7.5	(0.2)	0.24	(0.07)	1.31	(1.69)	0.19	(0.12)	7	(7)	2	(2)	115	(21)
TS8A	x				x		7.7	(0.2)	0.40	(0.08)	0.62	(0.57)	0.38	(0.17)	6	(3)	1	(0.4)	187	(26)
TS8B	x				x		7.7	(0.2)	0.37	(0.07)	1.02	(0.75)	0.79	(1.94)	6	(3)	2	(1)	170	(23)
TS9					x		7.6	(0.2)	0.28	(0.12)	0.88	(0.83)	0.29	(0.34)	5	(3)	1	(1)	98	(13)
TS10	x				x		7.7	(0.3)	0.42	(0.10)	0.32	(0.34)	0.64	(1.78)	5	(2)	1	(0.4)	188	(27)
TS11				x		x	8.1	(0.3)	4.78	(1.28)	0.44	(0.36)	0.46	(0.22)	8	(9)	2	(2)	605	(102)
TS12					x		7.6	(0.3)	0.31	(0.06)	0.91	(1.62)	0.54	(1.58)	5	(2)	1	(0.4)	124	(16)
TS13	x				x		7.4	(0.2)	0.53	(0.18)	0.93	(1.67)	0.63	(0.23)	10	(8)	2	(1)	201	(36)
TS14	x	x			x	x	7.4	(0.2)	0.42	(0.09)	2.99	(9.41)	0.39	(0.19)	16	(22)	3	(7)	175	(23)
TS15	x	x			x		6.7	(0.5)	0.45	(0.12)	0.84	(1.04)	1.45	(1.51)	23	(23)	1	(1)	201	(26)
TS16	x	x			x		4.6	(0.4)	0.92	(0.36)	1.28	(1.40)	6.38	(4.54)	276	(197)	7	(6)	463	(173)
TS17	x	x			x		5.5	(1.2)	0.77	(0.33)	5.44	(5.17)	2.60	(1.11)	169	(102)	9	(8)	289	(69)

Table I1b – Average Monthly Water Quality Data for 1996 to 1997; All Flow Conditions. Standard deviations are given in parenthesis. For all parameters except TSS n = 15 to 16; for TSS n = 20; for all reference sites n = 5. Urban use defined by Ohio small town influences or sanitary NPDES discharge; Nos.2 and 31 are active underground long-wall mines below major drainage; AUML - abandoned underground mine lands above major drainage; ASML - abandoned surface contour mine land; and AG - agricultural activities. LCS1 and sites below to a significantly lessor degree are also influenced by a closed quarry operation.

Station ID	ASML	AUML	No.2	No.31	AG	Urban	Ammonia-N (mg/L)		NO3+NO2 (mg/L)		Phosphorous (mg/L)		Chloride (mg/L)		Sulfate (mg/L)		TOC (mg/L)		TSS (mg/L)	
(R)FCS1	x	x			x		0.05	(0.00)	0.06	(0.02)	0.09	(0.18)	18	(14)	85	(21)	3.6	(2.1)	80	(171)
(R)SCS1	x	x			x		0.05	(0.00)	0.14	(0.11)	0.02	(0.02)	6.2	(1.9)	65	(5.0)	5.4	(2.8)	13	(6.3)
(R)SCS2	x	x			x		0.05	(0.00)	0.20	(0.18)	0.05	(0.05)	8.0	(2.0)	62	(7.8)	4.6	(2.3)	24	(22)
LCS1					x	x	0.06	(0.03)	0.28	(0.21)	0.04	(0.06)	6.3	(2.1)	86	(53)	4.3	(2.3)	13	(25)
LCS2					x		0.07	(0.05)	0.28	(0.17)	0.06	(0.09)	5.6	(1.9)	55	(22)	4.3	(2.4)	23	(46)
LCS3					x		0.05	(0.01)	0.29	(0.17)	0.05	(0.05)	5.7	(2.1)	47	(14)	4.4	(2.3)	27	(60)
LCS4			x		x		0.06	(0.03)	0.25	(0.13)	0.04	(0.05)	12	(8.3)	108	(102)	3.8	(1.9)	28	(43)
LCS5B	x		x		x		0.05	(0.01)	0.24	(0.11)	0.03	(0.03)	8.6	(3.3)	90	(38)	3.4	(1.5)	15	(10)
LCS6	x		x	x	x		0.27	(0.17)	0.36	(0.17)	0.07	(0.16)	110	(104)	673	(560)	3.4	(2.0)	35	(97)
LCS7	x		x	x	x		0.15	(0.11)	0.39	(0.23)	0.04	(0.06)	99	(92)	641	(542)	3.5	(2.0)	33	(85)
LCS8	x	x	x	x	x		0.12	(0.10)	0.42	(0.23)	0.03	(0.05)	100	(93)	603	(507)	3.2	(1.6)	27	(56)
LCS9	x	x	x	x	x		0.11	(0.09)	0.41	(0.21)	0.03	(0.04)	96	(89)	589	(464)	3.1	(1.5)	22	(20)
LCS10	x	x	x	x	x		0.11	(0.09)	0.36	(0.16)	0.05	(0.09)	68	(56)	452	(295)	3.3	(1.5)	48	(105)
TS1					x	x	0.51	(0.56)	0.42	(0.19)	0.17	(0.11)	11	(5.4)	37	(13)	5.7	(1.9)	15	(16)
TS2					x	x	0.21	(0.38)	0.44	(0.35)	0.06	(0.05)	9.4	(7.8)	52	(20)	4.5	(1.8)	17	(19)
TS3B					x		0.38	(0.49)	0.48	(0.29)	0.15	(0.10)	6.0	(3.3)	39	(12)	4.6	(2.1)	31	(43)
TS4					x		0.06	(0.03)	0.41	(0.29)	0.03	(0.03)	7.1	(2.1)	40	(13)	3.4	(1.5)	14	(37)
TS5					x		0.07	(0.05)	0.22	(0.11)	0.03	(0.03)	5.5	(1.4)	36	(9.3)	2.9	(1.5)	16	(30)
TS6			x		x	x	0.08	(0.09)	0.21	(0.10)	0.03	(0.03)	27	(28)	251	(278)	3.1	(1.7)	34	(54)
TS7					x		0.05	(0.00)	0.07	(0.04)	0.02	(0.02)	2.7	(1.9)	41	(13)	3.5	(1.7)	23	(31)
TS8A	x				x		0.05	(0.00)	0.13	(0.08)	0.01	(0.01)	5.8	(1.9)	123	(28)	2.9	(0.9)	15	(14)
TS8B	x				x		0.05	(0.00)	0.11	(0.05)	0.03	(0.03)	4.1	(1.2)	94	(20)	3.5	(1.4)	25	(28)
TS9					x		0.05	(0.00)	0.18	(0.10)	0.02	(0.02)	4.7	(1.4)	43	(7.7)	2.5	(0.8)	18	(31)
TS10	x				x		0.05	(0.00)	0.13	(0.09)	0.02	(0.01)	4.0	(1.8)	103	(19)	2.2	(0.9)	6	(4.6)
TS11				x		x	1.08	(0.40)	0.34	(0.08)	0.01	(0.00)	349	(93)	1940	(473)	2.1	(1.1)	10	(6.5)
TS12					x		0.05	(0.00)	0.12	(0.07)	0.01	(0.01)	9.1	(3.6)	38	(6.0)	2.7	(1.0)	23	(39)
TS13	x				x		0.05	(0.00)	0.15	(0.09)	0.02	(0.03)	31	(29)	140	(16)	2.3	(0.7)	29	(42)
TS14	x	x			x	x	0.05	(0.01)	0.34	(0.17)	0.10	(0.26)	9.2	(4.0)	121	(27)	3.1	(1.6)	13	(11)
TS15	x	x			x		0.05	(0.01)	0.21	(0.10)	0.02	(0.01)	3.5	(1.4)	178	(28)	2.1	(1.0)	34	(48)
TS16	x	x			x		0.08	(0.05)	0.14	(0.07)	0.01	(0.01)	3.4	(1.1)	553	(270)	2.3	(0.7)	12	(13)
TS17	x	x			x		0.12	(0.04)	0.24	(0.10)	0.03	(0.03)	24	(9.1)	321	(112)	2.4	(0.8)	37	(56)



(a)



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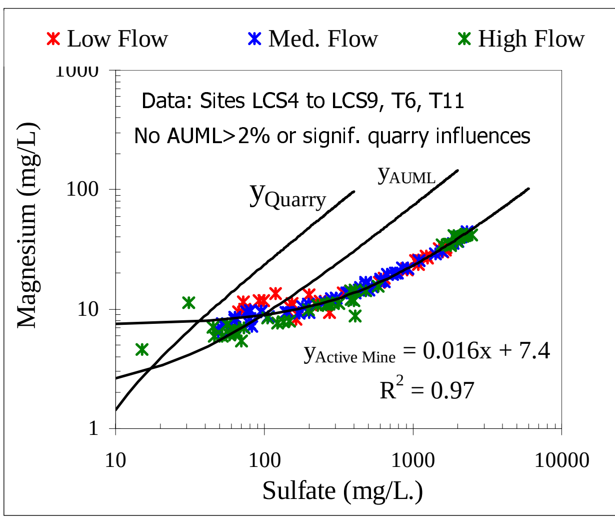

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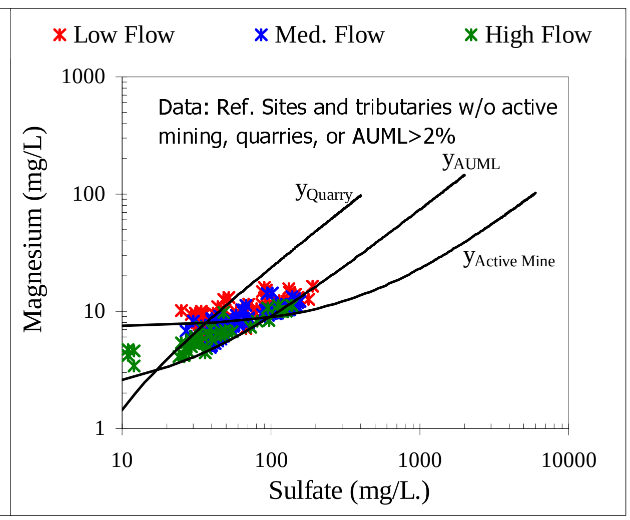

(d)

Figure I1 – Water Column Signatures for Total Magnesium and Sulfate: (a) Limestone Quarry Influenced, (b) Untreated Pyretic Acid Mine Drainage (AMD) from AUML, (c) Treated Pyretic AMD from Active Underground Mines, (d) Background With i) No Active Mine Discharges, ii) No Abandoned Underground Mine Influences, and iii) No Limestone Quarry Influences.

VITA

Justin Eric Babendreier

Justin Eric Babendreier graduated in 1984 from St. Mary's College of southern Maryland with a Bachelor of Science degree, majoring in Mathematics and Natural Sciences. Supported by the Pratt Fellowship, he went on to complete a Masters Degree in Control Systems Engineering within the Bradley Department of Electrical Engineering at Virginia Tech, graduating in 1987. He then worked as a consulting engineer in environmental areas between 1987 and 1994. Beginning with Olver Incorporated and then joining Draper Aden Associates, both located in Blacksburg, Virginia, his consulting work focused on multimedia projects and assisting various industries and government municipalities with regulatory compliance assistance. Project work entailed solid and hazardous waste management activities, groundwater and surface water investigations, and wastewater treatment systems and discharges.

Since 1994, Justin Eric Babendreier served as a private consulting engineer for various industries while completing Ph.D. studies in Civil and Environmental Engineering at Virginia Tech. During this period, he also continued to serve as Technical Services Advisor on environmental projects and Senior Project Manager for environmental permitting work at the firm Draper Aden Associates. He became a Professional Engineer in 1999. His academic awards include receipt of the first annual Jeanne Brocavich Memorial Mathematics Award at St. Mary's College and the St. Mary's Scholar award.