Effects of Surface Mining and Residential Land Use on Headwater Stream Biotic Integrity in the Eastern Kentucky Coalfield Region

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Effects of Surface Mining and Residential Land Use on Headwater Stream Biotic Integrity in the Eastern Kentucky Coalfield Region

by

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Executive Summary

The Kentucky Division of Water's (KDOW) surface water monitoring programs rely on biological, chemical, and habitat information to make science-based judgments on aquatic life use-support designations. This report documents biological impairment to macroinvertebrate communities in headwater streams primarily disturbed by surface coal mining and residential land use in eastern Kentucky. These two primary land uses are considered to be long-term and geographically pervasive throughout eastern Kentucky.

In order to assess waterbody health, KDOW compares stream data to reference conditions. The reference condition collectively refers to the range of quantifiable ecological elements (i.e., chemistry, habitat and biology) that are found in least-disturbed environments. KDOW has an extensive reference reach network (>200 sites) located throughout the Commonwealth. Nearly half of these streams are located in headwater watersheds. In the Eastern Coalfield Region (ECF), KDOW utilizes approximately 40 headwater reference sites to set criteria for aquatic life use designations. Headwater streams are important resources that serve multiple functions (e.g., water supply, waste assimilation, flood control and ecological values) often overlooked in environmental planning and land-use decision making. These sometimes intermittent waterbodies (primarily 1st and 2nd order streams) serve as the key interface between the surrounding landscape and larger waterbodies and provide high quality water for downstream uses (Yoder et al. 2000, Wallace and Meyer 2001).

Although state and federal regulatory requirements to protect water quality exist, impacts to streams due to surface mining are still common and widespread. Surface mining impacts streams both chemically and physically by increasing dissolved solids (e.g., sulfate, calcium carbonate) and sediment loading, and by removing riparian forest vegetation. Residential development in the ECF headwaters probably has the least oversight with regard to the protection of aquatic resources. Both home site construction and occupation within and adjacent to stream corridors can seriously impact aquatic species and their habitats through stream channelization and by increasing nutrients and organic wastes, sediment loads and removing riparian forest vegetation. Other impacts to ECF headwater streams arise from timber harvesting, road construction, oil and gas development, and light agriculture.

A total of 83 sites (38 reference sites and 45 disturbed sites) with watershed areas less than four square miles were sampled for macroinvertebrates and habitat and physicochemical parameters. These data were collected over a four-year period between 2000 and 2004, with the majority of the data collected between 2000 and 2002. Headwater streams were sampled between mid-February and late-May (spring index period). Sites were categorized *a priori* into one of four groups (reference, residential, mined/residential, and mined) based on the predominant land use upstream of the sampling reach. Macroinvertebrate sampling was conducted in accordance with *Methods for Assessing Biological Integrity of Surface Waters in Kentucky* (KDOW 2002). Physicochemical parameters (conductivity, pH, dissolved oxygen, and stream temperature) were collected and habitat features were scored with the EPA Rapid Bioassessment Protocol (RBP) Habitat Assessment procedure following Barbour et al. (1999). Stream canopy closure was also estimated in each reach and scored.

Sites were assessed with the Kentucky Macroinvertebrate Bioassessment Index (MBI), an aggregate index that incorporates seven metrics: 1) total generic taxa richness 2) total generic EPT richness; 3) the modified Hilsenhoff Biotic Index (mHBI); 4) the modified %EPT, which excludes the tolerant caddisfly *Cheumatopsyche*; 5) %Ephemeroptera abundance; 6) %Chironomidae+%Oligochaeta abundance; and 7) %Clingers. Exploratory box plots and scatter plots were viewed along with Pearson correlation coefficients and linear regression to evaluate relationships between environmental and biological data. Multivariate techniques included principal components analysis (PCA), stepwise discriminant function analysis (DFA), and correspondence analysis (CA). Significance tests were performed on environmental and biological parameters between the reference and the other three land-use categories with the non-parametric Kruskal-Wallis multiple comparison z-value (rank sum) test.

There were significant differences in conductivity ($p<0.05$) between most categories, but reference and residential sites were not significantly different. Reference sites averaged 63 μS/cm, while residential, mined/residential, and mined sites averaged 195, 552, and 1096 μS/cm, respectively. pH was significantly higher ($p<0.05$) at mined sites than at reference and residential sites. Reference and residential sites were not significantly different, nor was residential versus mined/residential sites. Reference sites averaged 6.7 while residential, mined/residential, and mined sites averaged 7.3, 7.8, and 8.0 S.U., respectively. Reference streams were significantly higher in total habitat scores ($p<0.05$) but no substantial differences were detected between residential, mined/residential, and mined sites. The average reference habitat score was 169 (out of 200), while residential, mined/residential, and mined sites averaged 136, 130, and 130, respectively. Canopy scores were significantly highest at reference sites ($p<0.05$). Mined/residential sites had significantly lower canopy scores than mined or residential sites.

The dispersion of disturbed sites in PCA ordination space clearly demonstrated that habitat and physicochemical factors deviated from the reference condition. The RPB total habitat score had the highest factor loadings on axis 1 followed by RBP epifaunal substrate score and conductivity. Taxonomically, visual inspection of the CA ordination suggested that reference communities were highly similar to each other. However, substantial departure of most residential and mined sites from the reference site array indicated very different community makeup. This analysis also demonstrated distinct separation of assemblages from mined sites versus residential sites. Mined/Residential streams were different from the reference site cluster and plotted fairly evenly throughout mined and residential clusters in ordination space.

Streams from the three disturbed categories had significantly lower MBI scores, taxa richness, EPT richness, m%EPT, %Ephem and %Clingers, and significantly higher mHBI and %Chir+Olig values than reference sites ($p<0.05$). The MBI and its associated metrics were significantly correlated ($p<0.05$) to conductivity, pH, RBP total habitat score, the RBP embeddedness score, epifaunal substrate score, and sediment deposition score, with correlation coefficients greater than $\pm$ 0.45. Between the three disturbed landuse categories, a slight pattern was detected graphically that distinguished effects of individual land uses from conductivity influences, but not habitat quality. The wholesale loss of mayflies (%Ephemeroptera and Ephemeroptera richness) at mined sites indicated that these organisms are especially sensitive to coal mine drainage. Dissolved solids emanating from hollowfills are a primary cause of biological impairment because of their severe impact to mayflies (a key component of headwater stream communities) and other sensitive taxa. Some residential sites produced similar harmful conditions for mayflies that may be linked to excessive nutrient and organic loading.

Overall, the MBI indicated nearly all (90-95%) of the streams with mined, residential, and mined/residential land use were impaired. The data presented here indicated that macroinvertebrate communities are extremely sensitive and vulnerable to the land uses that are most pervasive throughout eastern Kentucky (i.e., mining and residential). Both mining and residential impacts are unquestionably long term (decades or centuries), and are not likely to be eliminated by current regulatory efforts.

1.0 Introduction

The Kentucky Division of Water (KDOW) in the Department for Environmental Protection (DEP) is responsible for monitoring and assessing the ecological health of waterbodies across the Commonwealth. The primary focus of the Water Quality Branch of KDOW is upon surface waters ranging from small headwater streams to large rivers. Various land uses in the Commonwealth impart multiple impacts from both point and nonpoint sources to aquatic resources. This report documents biological impairment in headwater streams primarily disturbed by surface coal mining and residential land use in eastern Kentucky. These two primary land uses are long-term and geographically pervasive throughout eastern Kentucky. The heavily dissected Appalachian Plateau in eastern Kentucky (referred to hereafter as the Eastern Coalfields, or ECF) is drained by thousands of small headwater streams that ultimately feed hundreds of mid-sized streams and five major rivers (Big Sandy, Little Sandy, Licking, Kentucky and Upper Cumberland).

KDOW's surface water monitoring programs rely on biological, chemical, and habitat information to make science-based judgments on aquatic life use-support designations. Biological communities integrate and reflect environmental conditions; therefore, benthic invertebrate data were collected to further investigate degrees of impacts indicated by differences in water-quality at headwater sites scattered throughout the ECF. Although the KDOW routinely integrates three biological assemblages (algae, macroinvertebrates, and fish) in their water quality assessments, this investigation focuses only on macroinvertebrates. Benthic invertebrate populations are critical elements of aquatic food webs, and they possess many characteristics that make them good indicators of instream conditions (Cairns and Pratt 1993, Resh and Jackson 1993). Moreover, the use of invertebrate indicators in headwater streams is preferred since many of these streams have naturally depauperate fish communities because of their small size and steep gradients.

1.1 Importance of Headwater Streams

Headwater streams serve multiple functions (e.g., water supply, waste assimilation, flood control and ecological values) often overlooked in environmental planning and land use decision making. These often intermittent waterbodies (primarily 1st and 2nd order streams) serve as the key interface between the surrounding landscape and larger waterbodies and provide high quality water for downstream uses (Yoder et al. 2000, Wallace and Meyer 2001). Small 1st-2nd order streams represent the majority of shoreline within any drainage network and make up 86% of total stream length in the U.S. (Leopold et al. 1964). Because these small streams are so closely connected to their watersheds (Hynes 1975), terrestrial disturbances can result in severe and enduring impacts, which ultimately can affect both local and downstream environments (Webster et al. 1992). Brinson (1993) argued that the highest priority should be placed on protection of headwater streams and wetlands because of their close proximity to the surrounding landscape causes pronounced impacts cumulatively affecting downstream water quality.

In general, natural headwater streams in the ECF are narrow, shallow, cool, heavily shaded, and low in nutrients and dissolved ions. They are predominately heterotrophic, where energy is derived from allochthonous organic material provided by riparian vegetation (e.g., leaves, sticks and large woody debris). In contrast to larger wadeable streams and rivers, headwater streams are most susceptible to pollutant loading, having lower capacity for pollutant dilution and assimilation. The basic chemical composition of unpolluted streams draining a landscape is largely established and controlled in headwater streams (Gibbs 1970, Likens 1999, Johnson et al. 2000). Biotic uptake by vegetation, transformation by microbes in soils, riparian zones, and streams, in the presence of available carbon is an important mechanism controlling export of nitrogen from watersheds (Hedin et al. 1998). Small streams in the network are the sites of the most active uptake and retention of dissolved nutrients (Alexander et al. 2000, Peterson et al. 2001). Moreover, KDOW has found that higher proportions of sensitive and vulnerable species occupy headwater streams (Pond et al. 2003, KDOW unpub. data). Morse et al. (1997) stated that the Appalachian Mountains harbor many sensitive macroinvertebrates such as insects of the orders Ephemeroptera, Plecoptera and Trichoptera (EPT). The many endemic and rare species have been attributed to the diverse geological, climatological, and hydrological features of the region.

1.2 Reference Conditions

In order to characterize stream community health, biological, chemical, and physical data are compared to conditions found at reference streams (Hughes 1995). Reference streams are those that are least-impacted for a given geographic region, and theoretically should support rich and diverse communities with many pollution sensitive species and fewer tolerant species. Pond et al. (2000, 2003), and Pond and McMurray (2002) reported that the ECF region was relatively homogeneous with respect to the types of macroinvertebrate communities found at reference sites. Therefore, KDOW classifies the region collectively as the Mountain Bioregion. This region encompasses portions of three Level III ecoregions (Southwestern Appalachians [69], Central Appalachians [69] and the Western Allegheny Plateau [70] after Woods et al. [2002]). By comparison, the Mountain Bioregion (synonymous with the ECF region) macroinvertebrate fauna is distinct from other Kentucky bioregions (e.g., Bluegrass, Pennyroyal, Mississippi Valley-Interior Lowlands). Figure 1 shows an example of a headwater reference site in the ECF.

Figure 1. A headwater reference stream in Breathitt County.

Acknowledging past and present environmental stresses is important when considering reference sites. Natural disturbances from floods, droughts, windstorms, landslides, fires, and other phenomena were present in eastern Kentucky long before humans began to change the landscape. These disturbances, while either catastrophic or benign, have helped to shape the natural, expected aquatic community (Poff and Ward 1989). The hydrological conditions of forested watersheds in Kentucky were severely altered by logging in the early 1900s. Undoubtedly, severe erosion carried vast amounts of sediment into the stream channels. Many streams were used for log transport where channels were altered by the removal of obstructions such as large boulders and logs, and splash dams were built to provide high flows for transporting logs downstream. Evidence of this practice still remains in many streams in University of Kentucky's Robinson Forest where KDOW has several reference sites. Moreover, the once-forested steep slopes were often tilled for row crop agriculture after initial clear-cutting in the early 1900s, and small streams were frequently moved to one side of their valleys to accommodate farming or home building in the bottomlands. While there are no scientific data on the effects of these widespread activities, one can only speculate about the profound impacts to small headwater streams. In light of these historical impacts, reference streams selected by KDOW still represent the least-disturbed condition.

The reference condition collectively refers to the range of quantifiable ecological elements (i.e., chemistry, habitat and biology) that are found in least-disturbed stream environments. In Kentucky, finding reference streams can be a difficult task, because no regions are entirely without areas of some human disturbance. Ultimately, the application of the reference condition involves its comparison to streams exposed to various levels of environmental stress using defined sampling methodology and assessment criteria (KDOW 2002a). Impairment would be detected if indicator measurements (e.g., biological indices, habitat rating, chemical concentrations) fall outside the range of threshold criteria established by the reference condition (i.e., deviation below the reference distribution).

1.3 Primary Land-use Disturbances in the ECF

Stressors arising from mining, silviculture, residential and commercial development, agriculture, and road, railroad and bridge construction primarily affect watersheds in the ECF region. In the northern- and western-most parts of the ECF region (subecoregions 70h and 70b [after Woods et al. 2002]), agricultural impacts (e.g., livestock, row cropping, conversion of forest to pastureland) are more common than in the southern and eastern-most portions (subecoregions 69d-e and 68c [after Woods et al. 2002]). In the heart of the ECF region, mining operations and residential development are most pronounced in smaller watersheds where headwater mountain streams are exposed to more direct and profound physical and chemical disturbances.

1.3.1 Surface Mining Impacts

Although state and federal regulatory requirements for point and nonpoint sources to protect water quality exist, impacts to streams due to surface mining are still common and widespread (KDOW 2004, KDOW 2002b, KDOW unpub. data). Episodic releases of solids stemming from blackwater spills (i.e., coal slurry) and general mine operation runoff or releases can cause harm to aquatic biota. Chronic detrimental releases such as high concentrations of dissolved ions (e.g., sulfate) from hollowfills and other acid and non-acid mine drainage can have longer lasting effects, curtailing re-colonization and recruitment of sensitive invertebrate populations.

Physical impacts of mining and associated road construction and use include sedimentation and removal of riparian vegetation and associated organic matter inputs. Minor to severe sedimentation can occur as “pulse” events (blackwater releases, construction or failure of instream sediment ponds, culverts, or bridges) or as “press” events (mine site runoff, continual road runoff, poor BMP implementation). Furthermore, soil compaction and the presence of impoundments, roads, bridges, and culverts can contribute to excess sediment loading through modification of the hydrological regime (e.g., flow impediments, channel scour, stream bank failure). Figures 2, 3, 5, and 6 show typical mined landscapes in various stages of activity.

Many studies have documented that streams receiving drainage from mined areas exhibit several characteristics not found in unmined watersheds: 1) altered water-quality conditions (Curtis 1973, Dyer 1982, Hren et al. 1984, U.S. EPA 2002a); 2) increased sediment loads (Parker and Carey 1980, Osterkamp et al. 1984); 3) increased hydrologic response time to storm events (Bryan and Hewlett 1981), 4) altered flow duration curves (USGS 2001b), and 5) altered or changed channel morphology (A. Parola, Univ. of Louisville, pers. comm). These changes in the physical and chemical properties of stream environments can affect benthic invertebrate community structure and composition (Bradfield 1986, Green et al. 2000, Fulk et al. 2003).

Figure 2. View of a large contour surface mine in Bell County.

Figure 3. Lower end of an active mine in Upper Pigeon Branch, Pike County. More than 75% of this 2 sq. mile watershed (2nd order) was disturbed by mining activities. The site was located approximately 500 m downstream of this view. Conductivity and nitrate concentrations were highly elevated.

Mining significantly alters the chemistry of aquatic environments (Curtis 1973, Branson and Batch 1972, Minear and Tschantz 1976, Dyer 1982, U.S. EPA 2002b, Hartman et al. 2004). Acid mine drainage (AMD) is not as common in the ECF as it once was as enforcement, newer technology and mining methods have mostly eliminated it. However, sulfates are produced during surface mining, often in the form of calcium, magnesium, and iron complexes. Following mining, calcium, magnesium, manganese, and sulfate concentrations increase in a systematic fashion with the passage of time (Curtis 1973). Sulfate and conductivity is probably the most useful chemical indicator of the condition of a stream in mined watersheds in the ECF (Rikard and Kunkle 1990), and its concentration reflects the extent of watershed disturbance. Following a period of sulfate generation at the onset of mining, long-term production of the substance continues from mined watersheds (Minear and Tschantz 1976). Generally, high specific conductivity and concentrations of dissolved solids and hardness result from leaching of salts from crushed overburden (U.S EPA 2002a). In valley fills, Wunsch et al (1996) found that water emanating from the fills was calcium–magnesium–sulfate type water resulting from pyrite oxidation and calcite dissolution along the groundwater flow path.

General coal mine drainage (CMD) often causes physical and chemical impacts to streams as a result of the precipitation of entrained metals and sulfate, which become unstable in solution (U.S. EPA 2002a). Iron and aluminum usually precipitate as hydroxides, forming orange or white sludge (i.e., "yellow boy") that coats stream substrates. In addition, most mined streams in the ECF have elevated calcium in solution (Dyer 1982), and if pH is sufficiently elevated, gypsum ($CaSO_4$) will also precipitate (U.S. EPA 2002a). These sludge-like materials smother the stream bottom, armoring the substrate, thus inhibiting the feeding and reproduction of stream organisms. In addition, other precipitants such as calcium carbonate ($CaCO_3$) increase dramatically in receiving streams during and after mining operations. This calcium carbonate precipitates as a hard, encrusting and cementing substance. This substance also coats stream substrates and makes them unsuitable for colonization by invertebrates. Figure 4 shows a leaf pack broken off of a woody debris dam that had solidified with $CaCO_3$ and ferrous oxide.

Figure 4. A leaf pack cemented by $CaCO_3$ and ferrous oxide. This piece was broken off a larger debris dam in a headwater stream in Martin Co. affected by surface mining. The conductivity was 2350 μS/cm and the pH was 9.13 S.U.

Little is known about how or if heavy metals from CMD are responsible for aquatic life impacts in the EFC. Much argument has been made stating that elevated water hardness from current mining technology helps to control the bioavailablity of most metals. What is not known is whether biological processes, either in biofilms or through ingestion and digestion of particles, can make these metals bioavailable by the organisms themselves. Although water column samples contain only small quantities of dissolved or total metals, the bottom sediments may contain considerable quantities (Chapman 1978), usually attributed to adsorption on streambed materials or co-precipitation with the oxides and hydroxides of aluminum, iron, and manganese. Metal toxicity is dependent on the availability of the dissolved metal to the affected organisms, the exposure duration, and a host of other parameters. In situations where both iron and manganese are elevated, blooms of filamentous bacteria (*Leptothrix*) may smother benthic habitats. Sheaths of this iron-depositing bacterium have been known to be deleterious to macroinvertebrates by causing substrate avoidance, food quality limitations, and toxicity (Wellnitz et al. 1994).

Although KDOW has few data on nutrient concentration below mined-only watersheds, there is some evidence that shows surface mining leads to elevated nutrient levels. Moreover, increased nuisance algal growth has been observed below mining operations (pers. obs). Data from KDOW and University of Louisville researchers (J. Jack unpub. data) revealed roughly an 800 percent increase in nitrate levels above background conditions, although total phosphorus concentrations rose only by 50-75 percent. The elevated nitrate levels likely stem from careless handling or erosion of nitrate compounds used for explosives (U.S. EPA 2002a), runoff from nitrogenous fertilizers used in reclamation activities, and increased nitrogen export from loss of surrounding forest vegetation (Golladay 1988, Arthur et al. 1998).

Figure 5. Eastern Kentucky hollowfill setting showing typical worksite for pond cleanout and removal activities.

Figure 6. Close-up view of pond cleanout activities. Note the flow coming out of the toe of the hollowfill. The conductivity in receiving stream was 1235 μS/cm .

1.3.2 Residential Impacts

Residential development in the ECF headwaters probably has the least oversight with regard to the protection of aquatic resources. Both home site construction and occupation within and adjacent to stream corridors can severely impact aquatic species and their habitats. In the ECF region, many people live along small headwater streams (Figure 7) generally because of topographic limitations. Many of these streams have undergone localized channel changes to accommodate roads and housing, thus directly modifying instream habitat and indirectly affecting the natural flow regime. Moreover, increases in the impervious surface area causes streams to be more flashy and susceptible to channel scouring. Nutrient loading from residential activities can increase filamentous algal productivity that can ultimately smother important benthic habitats and alter invertebrate community integrity. Moreover, increases in nutrient concentrations in these normally nutrient poor headwater streams may stimulate blooms of filamentous bacteria (e.g., *Sphaerotilus*) that negatively affect macroinvertebrates (Lemly 1998, 2000). Even more harmful, elevated ammonia concentrations from untreated organic wastes can reach acute levels toxic to many aquatic taxa.

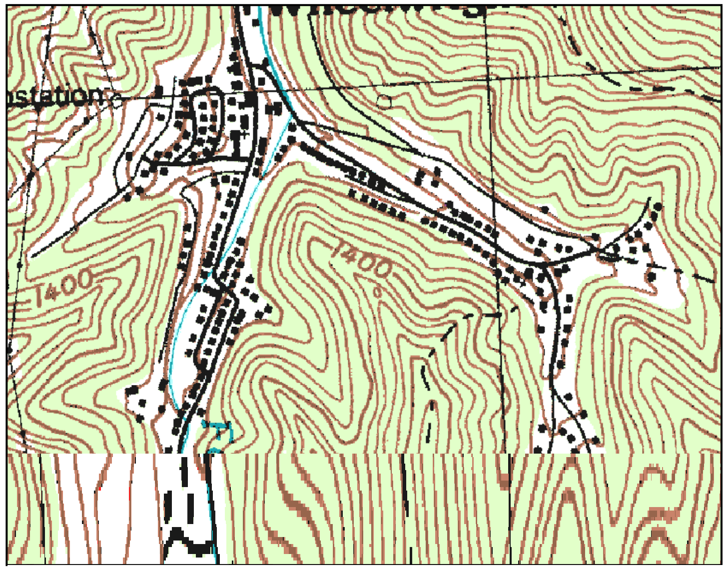

Figure 7. Close up view of 7.5 min topographic map in Floyd Co. showing high residential density along headwater streams.

Figure 8. View of sampling location downstream of map view shown in Figure 7 above.

The primary impact to aquatic organisms below unsewered high residential use areas stems from high organic loading from straight-pipe sewage and failing septic systems. Toxic, domestic household chemicals are also directly flushed into nearby streams. There are an estimated 15,000 straightpipes and an additional 15,000 failing septic systems in eastern Kentucky (KWRRI, 2002). Although efforts by eastern Kentucky PRIDE (a non-profit group) are being made to improve on-site wastewater treatment and provide sewer services in the ECF region, the magnitude of this problem will continue to have long-term impacts on water resources. Figures 9 and 10 highlight some typical problems associated with residential areas in the ECF.

Figure 9. View of a channelized headwater stream in a residential setting.

Figure 10. Solid waste accumulations in a wadeable stream in Floyd Co.

1.4 Other Types of Land-use Impacts

1.4.1 Logging Impacts

Timber harvesting can have profound effects on ecosystem-level processes, biological communities, and flow patterns (Webster et al. 1992, Likens et al. 1970). Most studies only address the impacts of clear-cutting; a method seldom practiced in Kentucky (Kentucky Division of Forestry, pers. comm.) except in site preparation for mining or highway projects (e.g., “clearing and grubbing”).

Soil disturbance associated with timber harvest can result in high sediment yields. Overland flow and erosion of mineral sediments into stream channels are promoted when these disturbances remove the litter layer or compact forest soils. This is often only a temporary impact since sediment yield decreases as natural vegetation reestablishes. However, severe erosion occurs when operators use stream channels illegally as skid trails (Figure 11). According to Leopold et al. (1964), stream channel morphology and associated habitats are directly influenced and maintained by eight major variables: channel width, depth, flow velocity, discharge, channel slope, roughness, sediment load, and sediment size. A change in any one of these variables will set in motion consequent changes in the other variables altering the structural attributes of the stream channel. The complex physical habitat that sustains resident biota depends upon the dynamic equilibrium sustained by the interaction of these variables. Thus, impacts associated with logging (or other watershed disturbance) that would change one or more of these variables such as the widening of channels for use as skid trails, changes in sediment load and sediment size due to changes in erosion patterns, increases in flow velocities due to the removal of canopy cover and floodplain roughness would cause channel instability and impair the stream's capacity to sustain complex physical habitat. The effects of such channel changes may be either long-term or short-term. In addition, by altering the flows of water and sediment, stream reaches above and below the impacted areas may be destabilized (Niemann et al. 2001, Tucker et al. 2001).

Figure 11. A headwater stream illegally used as a skid trail for logging operations.

Dissolved mineral loading may be increased slightly by harvesting but also declines quickly as vegetation reestablishes (Swank and Douglas 1977). Golladay (1988) and Arthur et al. (1998) found increases in nitrogen and phosphorus export in logged catchments in the Appalachians but minor differences in calcium, potassium, or sulfate concentrations between logged and undisturbed watersheds. Likens et al. (1970) actually found sulfate concentrations to decrease following clear cutting and experimental suppression of forest growth by herbicides.

Removing overhanging vegetation along heavily shaded headwater streams increases insolation, resulting in increased average temperatures. This is important since many headwater species have very narrow thermal tolerances. However, the duration of the temperature increase is typically short-lived (~5 years), with temperatures returning to pre-disturbance levels once the canopy closes over the stream (Swift 1983). Furthermore, removal of streamside vegetation results in a loss of allochthonous food material and may lead to stream bank stability problems.

1.4.2 Oil and Gas Impacts

The exploration and extraction of oil and gas reserves has also left a footprint on the landscape in the ECF region. There are more than 30,000 active and inactive wells in the ECF alone (Kentucky Geological Survey GIS layer). Prior to Kentucky's enactment and enforcement of stricter water regulations in the 1980's, brine wastes caused severe salinization of streams impairing many waterbodies. During the drilling and pumping process, brine water can migrate up through improperly cased wells and seep into nearby streams and groundwater. Elevated conductivity from chlorides (frequently > 10,000 μS/cm) can have dramatic effects on stream fishes, invertebrates, and algal communities (KDOW 1986, 1989, 1990). Fortunately, most of these streams have undergone complete recovery (KDOW unpub. data) following cessation of oil extraction or proper containment and disposal of brine water. Land disturbance associated with oil/gas production is generally minimal, but poor BMP implementation on access roads and at the worksite can increase sedimentation in nearby streams (pers. obs.). In some headwater areas, access roads are illegally located directly within the stream channel, causing severe sedimentation and habitat degradation.

1.4.3 Road Impacts

Roads that cross or lie near stream channels affect both the route and time in which storm water takes to reach the aquatic system. The results are intensified erosion, increased sediment loading, and changed runoff patterns. Reid and Dunne (1984) found that heavily used gravel roads contributed more than 100 times as much fine sediment as an abandoned road or a paved road. With paved roads, most of the sediment came from the associated ditches and cut slopes. Often, improperly placed and sized culverts contribute to local erosion and sedimentation both upstream and downstream of the crossings (Wellman et al. 2000, Warren and Pardew 1998).

There are few roadless watersheds in Kentucky. Numerous KDOW reference sites have gravel, paved, or grassed roads along their corridors. Indeed, some of these roads locally contribute noticeable amounts of sediments or cause stream bank stability problems. KDOW believes that using some reference sites with roads is acceptable and allows for a more realistic concept of attainability. The logic here is that most roads are permanent, and that total removal of roads from small watersheds, especially on private land, is impractical. However, roads and associated culverts or bridges placed with little or no oversight could damage stream channels to the point that the stream would not be considered reference quality.

2.0 Methods

2.1 General Study Area

The study region includes parts of the Southwestern Appalachian (68), Central Appalachian (69), and Western Allegheny Plateau (70) Level III ecoregions (Woods et al. 2002) in Kentucky (Figure 12a). All ecoregions lie within the Eastern Coalfield Physiographic Province (or Appalachian Plateaus Province), which makes up approximately 31% of the Commonwealth. This area is characterized by dissected terrain with similar forest types, geology, and climate. Bedrock geology is sedimentary and consists of interbedded sandstones, siltstones, shale, and coal and the dominant vegetation is part of the mixed mesophytic forest classification (Braun 1950). Common tree species found along reference streams include eastern hemlock, beech, maples, oaks, hickories, buckeye, and tulip tree. Common shrubs include spicebush, witch hazel, pawpaw, rhododendron, hydrangea, and ironwood. Headwater streams in this region typically flow through constrained valleys with high gradients and have boulder-cobble substrates. Precipitation patterns are generally uniform throughout the study region. In 1999, the summer prior to the onset of intensive data collection in headwater streams by KDOW, the eastern Kentucky region attained both severe and extreme drought status (Drought Mitigation Center 2001). Annually, the regional drought of 1999 fell near the 5th percentile for normal annual precipitation with a recurrence interval of more than 20 years (Institute for Water Resources 2001).

2.2 Site Selection

All biological, habitat, and chemical data used in these analyses are stored in KDOW's Ecological Data Application System (EDAS, v. 3.01) database. A total of 83 sites (38 reference sites and 45 disturbed sites) with watershed areas less than four square miles were used in this study (Figure 12b; Appendix A). These data were collected over a four-year period between 2000 and 2004, with the majority of the data collected between 2000 and 2002. Headwater streams were sampled between mid-February and late-May (spring index period). This is the period when macroinvertebrates are most diverse and abundant in headwater streams and therefore provides the most information for assessment purposes (Pond 2000, KDOW unpub. data).

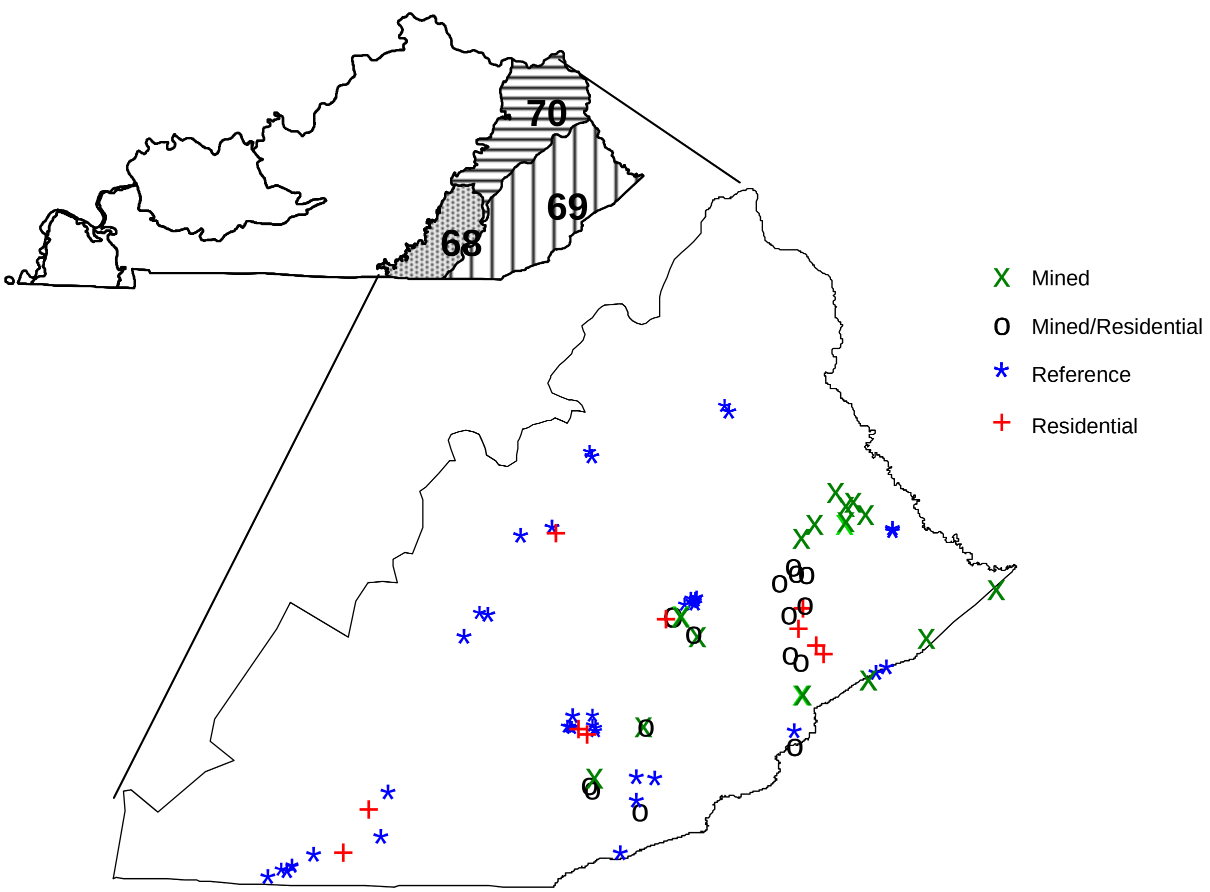

Figure 12. Level III Ecoregions that make up the ECF in Kentucky (a.), and location of sites (b.) coded by land-use category (see below).

Sites were categorized *a priori* into one of four groups (reference, residential, mined/residential, and mined) based on the predominant land use upstream of the sampling reach (Table 1). This categorization generally followed works by Green et al. (2000) and Fulk et al. (2003). To select reference headwater streams, intensive field and desktop reconnaissance was done using a combination of narrative and quantitative physical attributes (KDOW 2002a). Additional agency data were also reviewed (e.g., presence/absence of permitted dischargers, mines, oil and gas development and land cover) to help select candidate reference reaches. Biological data were not used to select reference sites to avoid circularity.

Table 1. Number of sites by category and mean and range (in parentheses) of catchment area (in sq. mi.).

	Total Number of Sites	Catchment Area
Reference	38	1.17 (0.13-3.15)
Residential	9	1.64 (0.3-3.3)
Mined	19	0.93 (0.8-3.7)
Mined/Residential	17	2.26 (0.1-2.81)

For the disturbed site categories, land use was determined using various agency data layers and Geographical Information System software (GIS, Arcview v. 3.1). Topographic maps, aerial photos (March 1995 and February 2002), mining map overlays, permitted dischargers, etc. were reviewed in relation to biological sampling points. Residential sites were included for analysis if >5 homes were situated upstream of the sample reach and had no other permitted activities. Mined/residential sites had some surface mining and more than five residences upstream of the sample point. Mined sites had some surface mining and no residences upstream. The majority of the sites had hollowfills located within their watersheds. Sites that could not be placed into a distinct category defined above were excluded from further analyses. Actual sampling reach selection was generally dependent upon accessibility, position in the watershed, and whether the site was representative of the stream as a whole. Several sites were randomly chosen as part of KDOW probabilistic monitoring program. Some residential and mined/residential sites were considered to have high housing density (>100 houses/mile) within the stream corridor. These same streams' uplands were often up to 80 percent covered in forest. In contrast, many mined sites had mostly forested stream corridors but very little forest in the uplands. In terms of streamside vegetation, residential and mined/residential streams were more likely to have non-native invasive species such as multiflora rose, Japanese honeysuckle, and Japanese knotweed along their banks. Representative sampling sites are shown in Figure 13.

With regard to oil/gas wells and roads, sites were retained for analysis if they had no more than two oil or gas wells upstream of the sampling reach (Kentucky Geological Survey GIS layer). A few reference sites had oil or gas wells in their watershed, but no elevated conductivity (an indicator of brines/chlorides) was detected. Roads of various usage were present in most watersheds and were assumed to contribute similar impacts among disturbed and reference categories.

Figure 13. Example sampling locations for the four land-use categories. Clockwise from top left: reference, mined, mined/residential, and residential.

2.3 Sampling Methodology

Macroinvertebrate sampling was conducted in accordance with *Methods for Assessing Biological Integrity of Surface Waters in Kentucky* (KDOW 2002a). Stream sites were typically assessed at the reach scale, generally 100 m in length. For all sites (reference and disturbed), it was impossible to assume that the available niches (e.g., stones in riffles, sticks in pools, leaf packs, fine sediments) were present in the same proportions; however, in nearly all streams, the same kinds of niches were available for sampling within the 100 m reach. Riffles were sampled semi-quantitatively using a kicknet or D-frame net. Four 0.25 m^2 samples were collected near riffle thalweg areas and composited to make a 1-m^2 sample. To eliminate effects of substrate diversity biasing the semi-quantitative sampling, an effort was made to sample riffle habitats that afforded macroinvertebrates with the best arrangement or layering of cobble, gravel, and small boulders (e.g., habitat complexity, availability). Non-riffle habitats were sampled qualitatively to try to collect as many species as possible within the stream reach. A summary of the collection methods is shown in Table 2. While this macroinvertebrate collection methodology is rather intensive, a sample was generally obtained within one hour. In the laboratory, all invertebrates were sorted from debris, identified to the lowest practicable taxon (usually genus or species level), and enumerated (except qualitative sample).

Table 2. Summary of sampling methods for headwater, moderate/high gradient streams.			
Technique	**Sampling Device**	**Habitat**	**Replicates (composited)**
$1m^2$ Kicknet* (quantitative)	Kicknet/Mesh Bucket	Riffle	4-0.25m^2 (total area= 1 m^2)
Sweep Sample (multi-habitat)	Dipnet/Mesh Bucket	All Applicable	
-Undercut Banks/Roots	Dipnet/Mesh Bucket		3
-Sticks/Wood	Dipnet/Mesh Bucket		3
-Leaf Packs	Dipnet/Mesh Bucket	Riffle-Run-Pool	3
-Silt,Sand, Fine Gravel	Dipnet/Mesh Bucket	Margins	3
Rock Pick	Forceps/Mesh Bucket	Pool	5 boulders
Wood Sample	Forceps/Mesh Bucket	Riffle-Run-Pool	2 linear m

*Sample contents kept separate from other habitats.

Physicochemical parameters (conductivity, pH, dissolved oxygen, and stream temperature) were collected using a portable Hydrolab® meter (Hydrolab Corp., Austin, Tex.). Habitat features were scored with the EPA Rapid Bioassessment Protocol (RBP) Habitat Assessment procedure following Barbour et al. (1999). This latter procedure qualitatively evaluates important habitat components such as epifaunal substrate quantity and quality, embeddedness, velocity/depth regimes, sediment deposition, channel flow status and channel alteration, stream bank stability, bank vegetation protection, and riparian zone width. Each component was scored on a 20-point scale with a total possible summed score of 200. For individual metrics and the total score, higher scores indicate better habitat and lower scores indicate habitat degradation. Stream canopy closure was also estimated in each reach and scored on an ordinal scale (1= 0-25%, 2= 25-50%, 3= 50-75%, 4= 75-100%).

3.0 Data Analysis

A combination of univariate, bivariate, and multivariate statistics were used to evaluate differences in a subset of environmental and biological parameters among the reference and three land-use categories. A previous KDOW study in the ECF (Pond and McMurray 2002) presented several distinct environmental and biological relationships that are re-emphasized here. The U.S. Army Corps of Engineers (USACE) has developed a rapid stream assessment protocol to estimate the ecological integrity of headwater streams in the ECF (Sparks et al. 2003a) by modeling many of the same parameters considered by Pond and McMurray (2002). The USACE model generates a similarity index that compares certain important abiotic and biotic ecosystem components of the stream to the conditions characterizing least disturbed headwater streams in the ECF region. The derived ecological integrity index is used in the context of the USACE regulatory program to help identify ways to avoid, minimize and compensate for any adverse impacts to aquatic functions and associated goods and services provided by these aquatic resources that may be at stake for projects seeking authorization under Section 404 of the Clean Water Act (Sparks et al 2003b).

Biological assessments were made with the Macroinvertebrate Bioassessment Index (MBI) and its associated metrics (Pond and McMurray 2002, Pond et al. 2003). Multimetric indices are used throughout the U.S. to assess waterbody health (Karr et al. 1986, Gerritsen 1995, Barbour et al.

1999, Karr and Chu 1999). The Kentucky MBI uses seven equally weighted metrics that are standardized to the 95th percentile of the reference data set. This standardization not only excludes outliers from the data set, but also allows for the combination of abundance and richness metrics. After standardization, metric scores are averaged to produce the MBI score on a 100-point scale. Effort was given to evaluate metrics covering a wide scope of ecological attributes (e.g., structure, tolerance, habit, and function) for desirable attributes such as sensitivity, lack of redundancy, correlation to stressors, and use compatibility with historical KDOW assessments and U.S. EPA guidance (e.g., Barbour et al. 1999).

The MBI's seven headwater metrics are:

1) total generic taxa richness (TR; increases with higher water quality);
2) total generic EPT richness (EPT; increases with higher water quality);
3) modified Hilsenhoff Biotic Index (mHBI), an abundance-weighted community tolerance metric on a scale of 0-10 (higher scores indicate increasing water quality degradation);
4) modified %EPT abundance (m%EPT; increases with higher water quality) which excludes the tolerant caddisfly *Cheumatopsyche*;
5) %Ephemeroptera abundance (mayflies; increases with higher water quality);
6) %Chironomidae+%Oligochaeta abundance (midges and worms; decreases with higher water quality); and
7) %Clingers abundance (taxa adapted to "cling" to stable substrates; increases with higher water quality).

Detailed descriptions for these metrics are provided in KDOW (2002), or Pond et al. (2003). State agencies in Tennessee (Arnwine and Denton 2001) and West Virginia (Gerritsen et al. 2000) also use combinations of these metrics in their bioassessment programs.

The MBI is broken down into five narrative water quality ratings. Excellent communities are those that score at or above the 50th percentile of the reference distribution. Good communities score between the 5th and 50th percentile. Trisection of scores below the 5th percentile yields narrative ratings of Fair, Poor, and Very Poor. Actual rating criteria are listed in Pond et al. (2003). For the purpose of this report, headwater MBI values below a score of 72 would be impaired (i.e., fair, poor and very poor).

Exploratory box plots and scatter plots were viewed along with Pearson correlation coefficients and linear regression to evaluate relationships between environmental and biological data. Multivariate techniques (i.e., non-testable, exploratory statistics) included forms of ordination: principal components analysis (PCA), stepwise discriminant function analysis (DFA), and correspondence analysis (CA). Ordination uses various algorithms that order sets of data points with respect to one or more axes (i.e., "the displaying of a swarm of data points in a two or three-dimensional coordinate frame so as to make the relationships among the points in many-dimensional space visible on inspection" [Pielou 1984]). To assure statistical normality for these multivariate techniques, physical and biological variables were transformed ($\log_{(x+1)}$, square root, or arcsine), where appropriate.

Species composition and abundance were evaluated with correspondence analysis (CA) using the statistical software package MVSP (Kovach Computing, London). Correspondence analysis is a weighted-average method that reciprocally double-transforms community data and computes eigananalysis to construct corresponding species and site ordinations (Ludwig and Reynolds 1988). CA was used for exploratory purposes in investigating how communities (genus-level sample data) differed from one another among land-use categories. In CA, sites are plotted as points along the first two axes (indirect environmental gradients) in species space. Points close together in ordination space indicate more similar faunal composition than points distant in ordination space.

Other multivariate techniques included principal component analysis (MVSP, Kovach Computing, London) and stepwise discriminant function analysis (DFA, SYSTAT v. 7.0). The former technique was used to elucidate patterns in abiotic factors related to individual sites and among *a priori* land-use categories. PCA also uses eigenanalysis and constructs orthogonal axes (components) where sites are plotted as points in ordination space, and environmental variables are plotted as vectors where their length and direction (correlations or loadings) depends on their statistical importance to the overall ordination. Stepwise DFA was used to select a subset of biological and habitat metrics, as well as physicochemical parameters that could best distinguish between the four land-use categories. Computationally, DFA is very similar to analysis of variance where the *F*-statistic is essentially computed as the ratio of the between-groups variance in the data over the pooled (average) within-group variance. A stepwise procedure “builds” the discriminant model with variables that can optimally differentiate between groups (i.e., land-use categories), while discarding less significant or autocorrelated variables.

Finally, significance tests were performed on environmental and biological parameters between the reference and the other three land-use categories with the non-parametric Kruskal-Wallis multiple comparison z-value (rank sum) test. This test was used to determine the significant differences between group means in an analysis of variance setting, with alpha set at 0.05.

4.0 Results and Discussion

4.1. Physical Comparisons

Environmental variables that are modified by watershed disturbance such as conductivity and sedimentation are well documented elsewhere in the literature (Branson and Batch 1972, Curtis 1973, Talak 1977, Dyer 1982, Green et al. 2000, Howard et al. 2001, USGS 2001a). Pond and McMurray (2002) reported that conductivity, sedimentation, and general habitat degradation were the most significant factors found between reference and impaired sites in ECF headwater streams.

In the present study, there were significant differences in conductivity ($p<0.05$) between most categories (Figure 14a). Reference and residential sites were not significantly different. Reference sites averaged 63 μS/cm, while residential, mined/residential, and mined sites averaged 195, 552, and 1096 μS/cm, respectively. The highest values were found at four mined sites where conductivity ranged between 1980 and 2490 μS/cm. It was apparent that sites with mining in their watersheds were contributing higher loads of dissolved solids. Green et al. (2000) also reported that the hollowfilled sites generally had comparable or higher conductivity than the filled/residential sites within a watershed, indicating that the probable cause of the increase in the conductivity at the filled/residential sites was the upstream mining activity rather than the residences. Natural stream chemistry in small streams in this region is often low in dissolved ions and has slightly acidic to circumneutral pH (Dyer 1982, Arthur et al. 1998). It is generally known that watershed disturbance and associated erosion increase streamwater ionic concentrations and subsequently conductivity (Curtis 1973, Dyer 1982, Dow and Zampella 2000). In general, runoff from coal mining operations (particularly mining practices that place overburden into hollowfills or valleyfills) contributes to this elevated conductivity and can add high amounts of sediment to receiving streams. As of 2002, approximately 730 miles of streams have been permanently buried by these practices in Kentucky (U.S. EPA 2002a). However, this figure takes into consideration only those blue-line streams that are shown on USGS 1:24000 scale topographic maps. Hundreds of miles of other headwater streams not shown on these maps have likely been filled.

pH was significantly higher ($p<0.05$) at mined sites than at reference and residential sites (Figure 14b). Reference and residential sites were not significantly different, nor was residential versus mined/residential sites. Reference sites averaged 6.7 while residential, mined/residential, and mined sites averaged 7.3, 7.8, and 8.0 S.U., respectively. Streams affected by extremely low pH from AMD (generally abandoned mines or underground works) are not as common as those affected by alkaline mine drainage. Substantial buffering of AMD occurs, in part, in response to the blending of semi-calcareous overburden in fills and reclaimed slopes. A study by Eastern Kentucky University (1975) concluded, “Alkaline pollution caused by surface mining is as real as acid mine drainage pollution.” Curtis (1973) and Dyer (1982) also documented this occurrence.

In terms of sedimentation and general habitat degradation, reference streams had significantly higher total habitat scores ($p<0.05$) but no substantial differences were detected between residential, mined/residential, and mined sites (Figure 14c). The average reference habitat score was 169, while residential, mined/residential, and mined sites averaged 136, 130, and 130, respectively. The embeddedness score (a measure of coarse riffle substrates covered in fine sediment) showed the greatest difference between reference and other categories ($p<0.05$), but no significant differences were detected between residential, mined/residential, and mined sites (Figure 14d). Sediment pollution from nonpoint sources is a serious problem in Kentucky (KDOW 2004, KDOW 2002b) and elsewhere (see Waters 1995). Small streams in the study area that have been exposed to mining and logging are subject to high sediment loading. Moreover, intensified bank erosion caused by hydrologic modification (e.g., impoundment, roads, bridges, and culverts) can substantially increase sedimentation in these streams.

Other factors such as reduced canopy cover and riparian width can have direct influences on macroinvertebrate communities that respond to stream temperature, bank habitat and stability, and changes in the food-energy base (e.g., Sweeney 1993). KDOW reference sites frequently had the natural complement of mature forest with dense canopies, albeit second-growth, but this condition was met at very few of the impacted sites. In intermittent streams, many aquatic insect taxa are adapted to resist desiccation through resting or diapausing eggs, larvae or pupae (Williams 1996). Dense summer canopies may maintain high relative humidity and reduce desiccation stress in the dry streambed sediments (Fritz and Dodds 2004), thus assuring recruitment of the next year's insect community. With regard to riparian zone width scores, reference sites had significantly higher scores than the three disturbed categories, and mined/residential sites had significantly lower scores than mined sites.

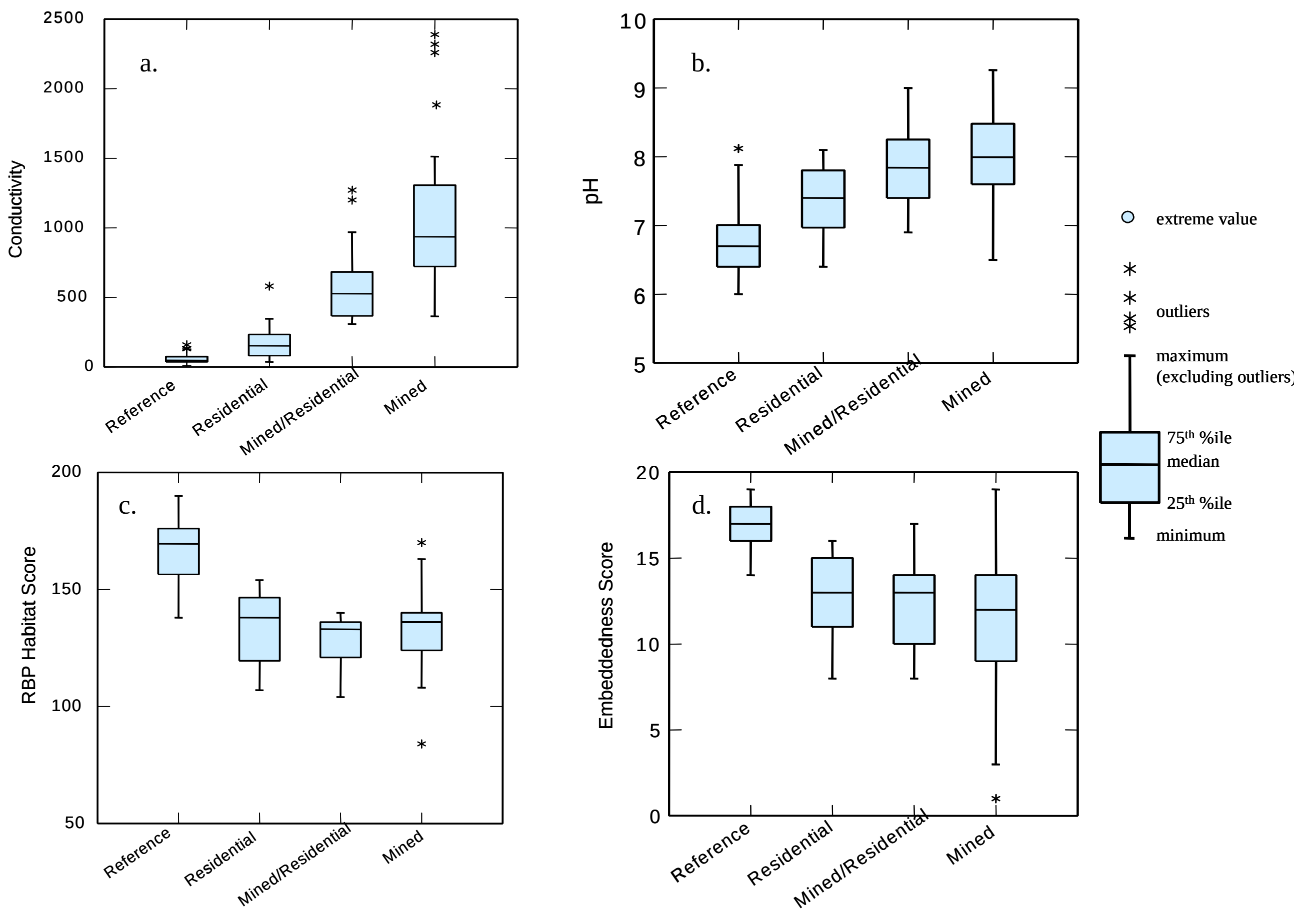

Figure 14. Box plots of (a.) conductivity (μS/cm), (b.) pH, (c.) Total RBP habitat scores, and (d.) RBP embeddedness scores among primary land-use types. Legend for box plots shown at far right.

The PCA ordination (Figure 15) verified that reference sites were highly similar with respect to physical variables such as RBP habitat parameters and physicochemical measurements. The dispersion of disturbed sites in ordination space also clearly demonstrated that physical habitat was different from the reference condition. It was not surprising that habitat metric scores (shown as arrows) were weighted toward reference sites in ordination space since by definition all reference sites have good habitat. The conductivity and pH vectors pointed toward impacted sites. Axis 1 explained 40.3 percent of the variance where axis 2 explained only 11.5 percent of the variance. Eigenvalues for the first four axes and PCA loadings (correlations) of all variables are shown in Table 3. The RPB total habitat score had the highest factor loadings on axis 1 (-0.39) followed by epifaunal substrate score and conductivity (-0.34 and 0.31, respectively). These parameters represent the most important factors related to the dispersion of sites along the horizontal axis.

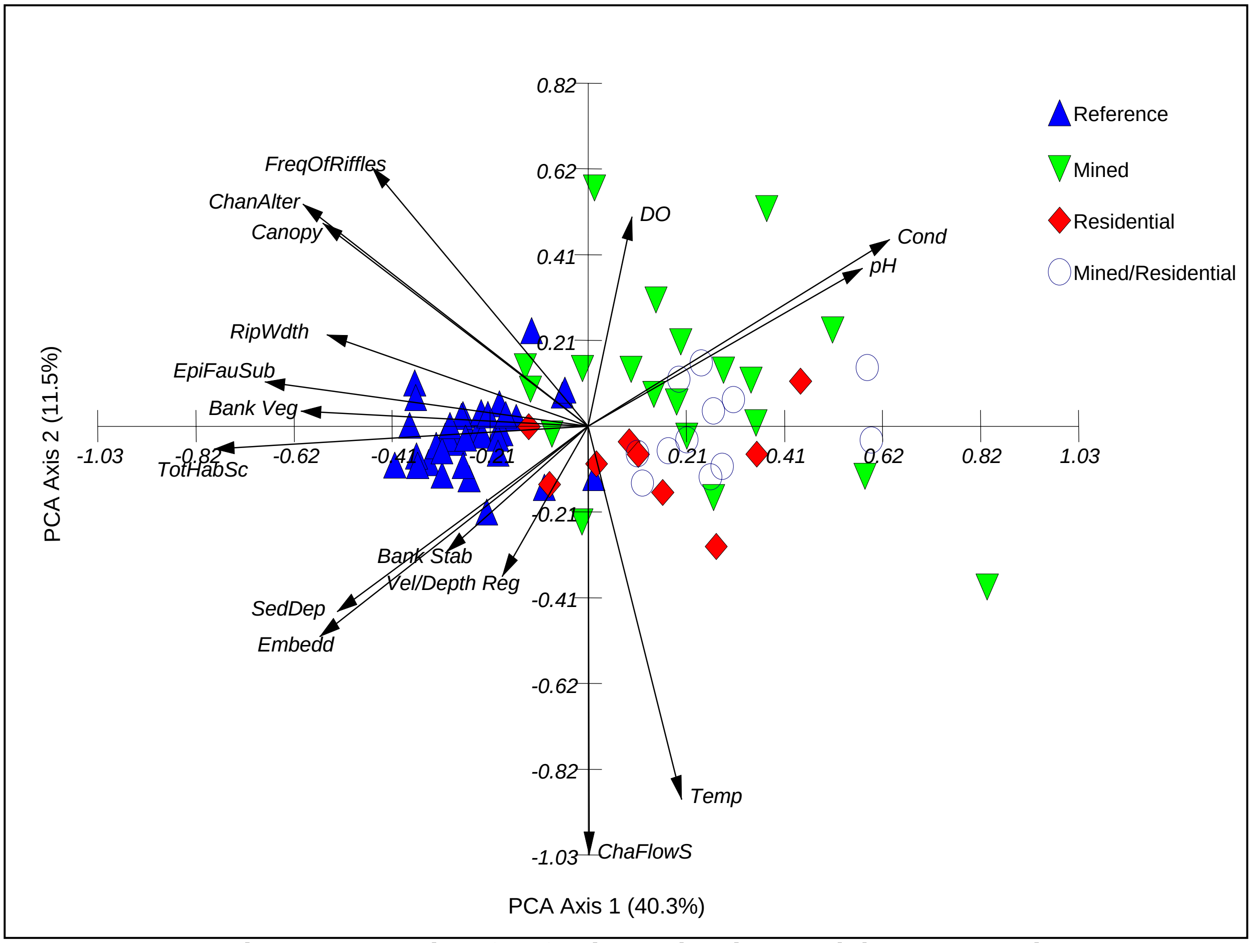

Figure 15. Principal components analysis (PCA) ordination based on RBP habitat metrics and physicochemical measurements (vectors) among land-use categories. Numbers in parentheses refer to the percent of the variance explained by each axis.

Temperature, channel flow status score, and frequency of riffles score had highest loadings on axis 2 (-0.44, -0.41, and 0.31, respectively) and caused a few sites to plot in outlying quadrants of the ordination (Figure 15). On axis 3 (8.3% variance explained, not plotted), bank stability score, bank vegetation protection score, and velocity/depth regime score had the highest factor loadings (Table 3). Dissolved oxygen, sediment deposition score, temperature, and canopy cover had the highest correlations (0.68, 0.37, 0.34, and –0.30, respectively) to axis 4 (7.2% variance explained, not plotted). Although axis 3 and 4 variables contributed much less than the first two axes, they added a combined 15.5% of the total explained variance. Compared to environmental conditions found at reference sites, the PCA ordination showed most of the mined sites had higher axis 1 and axis 2 coordinates, while the majority of residential sites plotted with higher axis 1 and lower axis 2 coordinates. This suggests measurable differences in these two land-use categories.

Table 3. Principle component analysis results for the first four axes for physicochemical data and RBP Habitat scores using all sites.

Eigenvalues	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	6.45	1.84	1.33	1.15
Percentage	40.31	11.47	8.32	7.17
Cum. Percentage	40.31	51.78	60.10	67.26

PCA variable loadings	Axis 1	Axis 2	Axis 3	Axis 4
Dissolved Oxygen (mg/L)	0.05	0.25	0.01	0.68
pH (S.U.)	0.28	0.19	0.20	-0.10
Temperature (centigrade)	0.10	-0.44	-0.18	-0.34
Conductivity (µS/cm)	0.31	0.22	0.13	-0.02
Bank Stability Score	-0.15	-0.15	0.59	0.14
Bank Vegetation Protection Score	-0.30	0.02	0.41	-0.07
Canopy Cover Score	-0.28	0.24	0.16	-0.30
Channel Flow Status Score	0.00	-0.41	0.21	0.06
Channel Alteration Score	-0.30	0.26	-0.14	-0.23
Embeddedness Score	-0.28	-0.25	-0.18	0.13
Epifaunal Substrate Score	-0.34	0.05	-0.16	0.11
Frequency of Riffles Score	-0.22	0.31	-0.27	0.08
Riparian Width Score	-0.27	0.11	0.20	-0.27
Sediment Deposotion Score	-0.26	-0.22	0.01	0.37
Velocity/Depth Regime Score	-0.09	-0.18	-0.37	-0.03
Total Habitat Score	-0.39	-0.03	0.03	0.01

4.2 Biological Considerations

4.2.1 Taxonomic Comparisons

Distinctive community level characteristics were found among the four land-use types. Visual inspection of the CA ordination (Figure 16) suggests that reference communities were highly similar to each other. There was considerable overlap among reference sites, indicating a relatively repeatable and predictable community in least-disturbed environments. Mined and residential sites that fell within the reference site cluster could possibly be considered unimpaired based on taxonomic composition and structure. These sites generally had lower conductivity and higher RBP habitat scores. However, substantial departure of most other residential and mined sites from the reference site array indicated very different community makeup. Mined/Residential streams were different from the reference site cluster and plotted fairly evenly throughout mined and residential clusters in ordination space. Although not analyzed further, it is important to note that mined/residential sites with less mining and more residential development plotted more closely with residential sites (negative portion of axis 2), while sites with more mining and less residential intensity plotted alongside mined sites (positive portion of axis 2). This further suggests disturbance-specific affinities by these invertebrate assemblages.

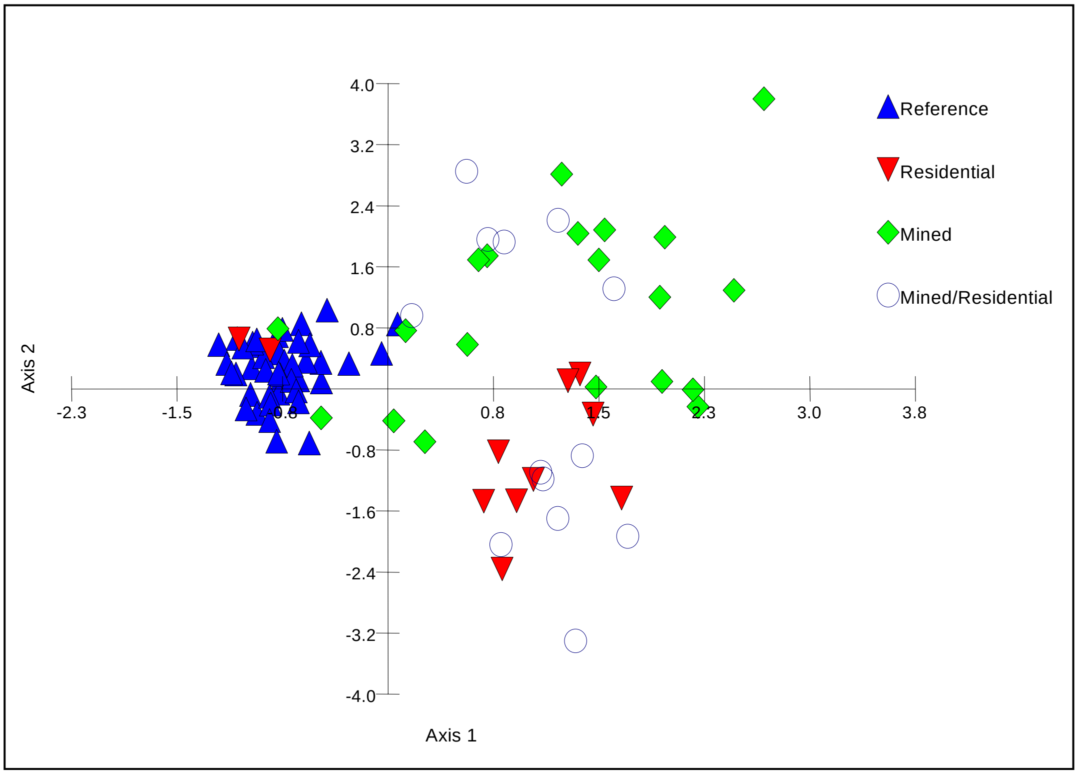

Figure 16. Correspondence Analysis of riffle-dwelling macroinvertebrate communities grouped by land-use category. Axis 1 and 2 explained 21% and 9% of the variance, respectively.

With regard to taxonomic composition, Figure 17 shows the occurrence frequency of the top 20 EPT taxa between reference sites and disturbed sites. While most genera were considered to be sensitive to disturbance, several taxa can be considered somewhat facultative to disturbance (e.g., genera occurring at >50% of the corresponding reference frequency for that taxon). For example, at mined sites the stonefly *Amphinemura* and caddisfly *Polycentropus* were frequently collected; at mined/residential sites, *Amphinemura*, the mayfly *Eurylophella*, and the stonefly *Isoperla* were fairly ubiquitous; and at residential sites, *Amphinemura*, *Eurylophella*, *Isoperla*, and the mayflies *Ameletus*, *Ephemerella*, and *Paraleptophlebia* were found fairly frequently.

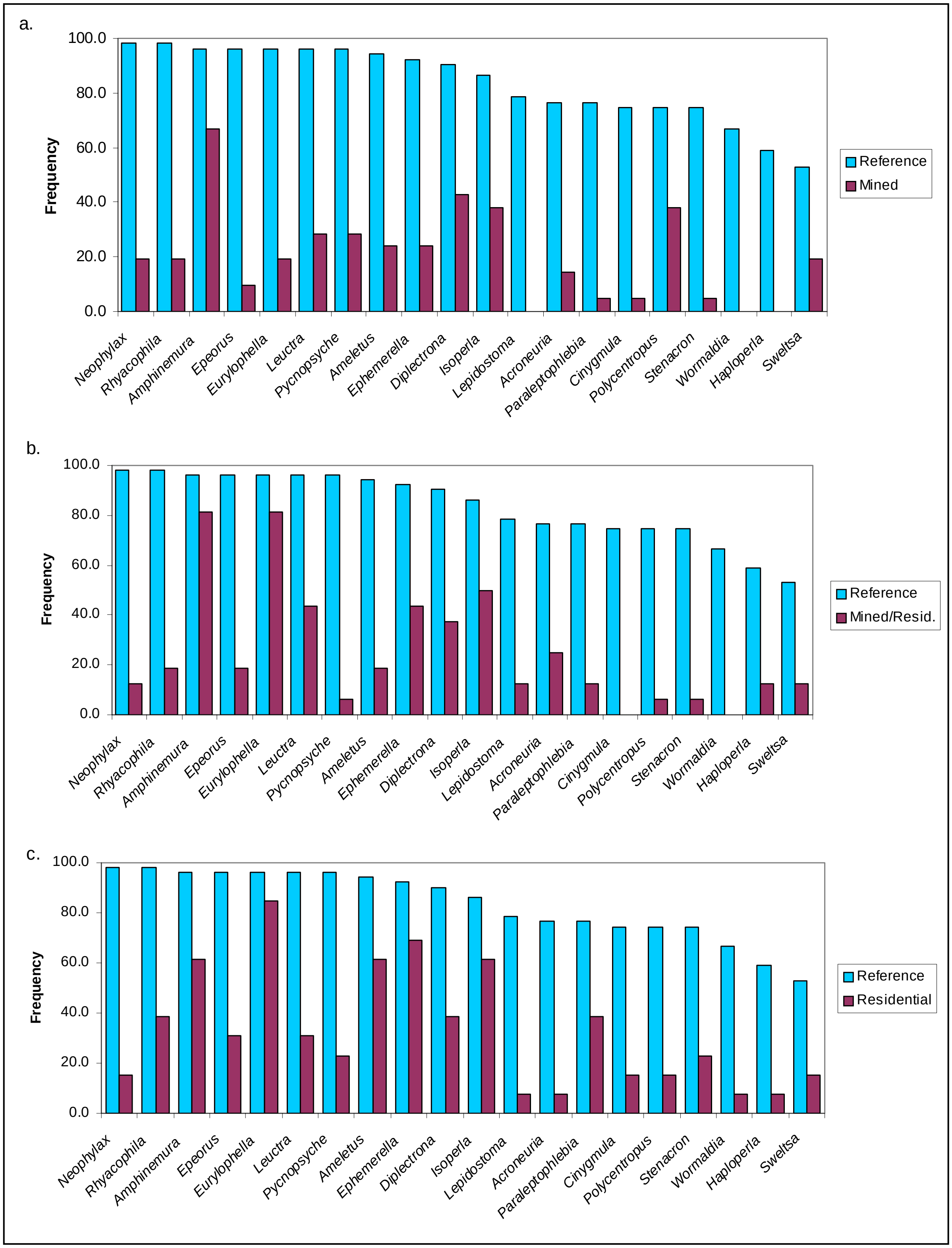

Figure 17. Presence/Absence frequency histogram comparing top most frequently collected EPT genera among reference sites versus (a.) mined, (b.) mined/residential, and (c.) residential sites.

4.2.2 *MBI and Metric Comparisons*

Table 4 shows MBI and metric values among the four land-use categories. Streams from the three disturbed categories had significantly lower MBI scores (also see Figure 18), taxa richness, EPT richness, m%EPT, %Ephem and %Clingers, and significantly higher mHBI and %Chir+Olig values than reference sites ($p<0.05$). There was considerable similarity between the three disturbed categories (Table 4 and Figure 19); however, the %Ephem was significantly reduced at mined sites compared to all other sites. Residential sites had the lowest m%EPT and mined/residential sites had the highest mHBI, %Chir+Olig, and the lowest %Clingers. Macroinvertebrate abundance was also affected, as the total number of individuals was significantly lower at mined and mined/residential sites than at reference and residential sites.

Table 4. Mean and range (in parentheses) of MBI scores, metric values, and total individuals (TNI; from quadrats) among four landuse categories in headwater streams in the ECF region. An asterisk (*) indicates significant difference ($p<0.05$) from reference; two asterisks (**) indicate significant difference from all other categories.

	MBI	TR	EPT	mHBI	m%EPT	%Ephem	%Chir+Olig	%Cling	TNI
Reference	84.6	48.9	26.9	2.56	77.2	48.9	3.6	61.7	543.9
	(69.2-95.6)	(27-64)	(19-36)	(1.67-3.14)	(50.1-96.4)	(17.3-73.3)	(0.1-11.6)	(28.5-82.9)	(110-1702)
Residential	42.1*	32.1*	11.7*	5.34*	25.4*	14.3*	43.1*	31.7*	442.9
	(7.1-74.6)	(18-45)	(0-20)	(2.09-8.41)	(0-97.1)	(0-51.5)	(0.1-99.6)	(0-61.6)	(114-1226)
Mined/Residential	39.7	31.1*	11*	5.47*	26.6*	13.3*	50.1*	24.3*	279.8*
	(15.4-82.3)	(10-45)	(1-25)	(3.41-6.85)	(0.6-80.2)	(0-56.9)	(4.3-95.5)	(5.1-57.4)	(9-658)
Mined	40.4*	31.1*	10.4*	5.25*	38.4*	4.1**	39.3*	29.1*	283.5*
	(19.8-91.2)	(16-42)	(5-19)	(3.05-6.87)	(9.1-84.1)	(0-34.2)	(4.8-79.1)	(4.2-69.4)	(85-514)

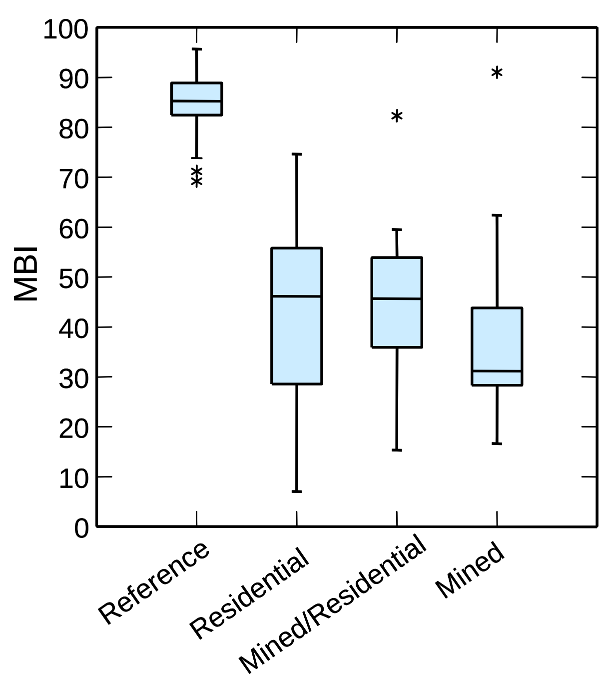

Figure 18. Boxplot of MBI scores among land-use types.

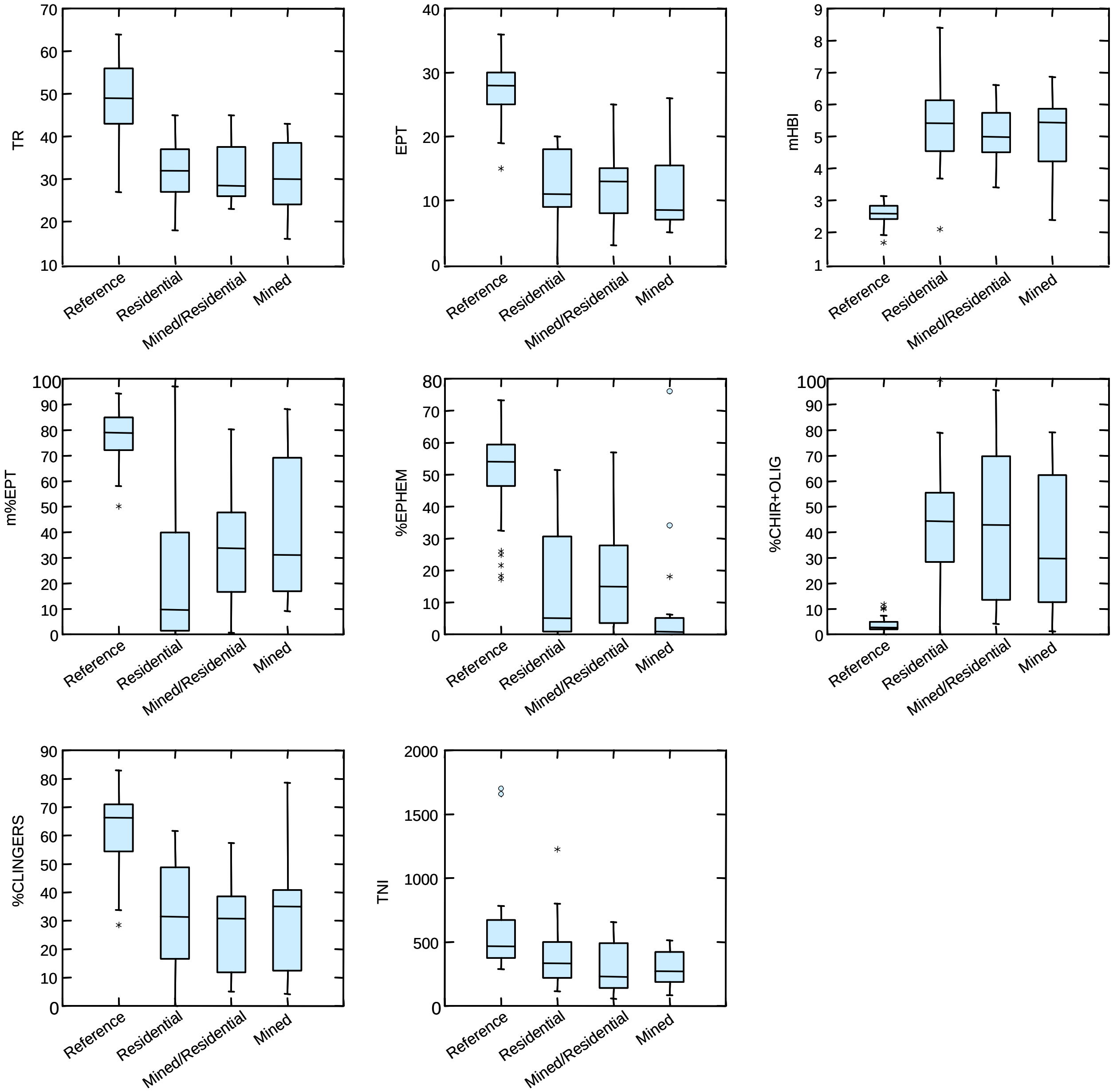

Figure 19. Box plots of MBI metrics and total number of individuals (TNI) among land-use categories.

The MBI and its associated metrics were significantly correlated ($p<0.05$) to many physicochemical parameters (Table 5). Namely, conductivity, pH, total RBP habitat score, the RBP embeddedness score, epifaunal substrate score, and sediment deposition score had the highest correlations to the MBI ($r > \pm 0.45$). Out of all of the RBP habitat metrics, embeddedness score had the highest correlation to MBI scores ($r = 0.64$) and all other MBI metrics (range of r =0.45-0.56). Some RBP metrics (i.e., bank stability, channel flow status, and frequency of riffles) were not significantly related to MBI scores or associated MBI metrics. No significant trend was found between the MBI and catchment area, suggesting that within the range of headwater watersheds used in this study (0.1–3.7 sq. miles), catchment area was not a factor. Furthermore, no significant differences were found among yearly reference site MBI scores (2000-2003, $p>0.05$). In an earlier KDOW study of ECF headwater streams, Pond and McMurray (2002) reported that several other parameters (i.e., dissolved oxygen, temperature, mean riffle substrate size, mean stream width) did not significantly correlate to the MBI or its associated metrics.

In a study of mountaintop removal mining and valleyfill impacts in West Virginia (Green et al. 2000), total taxa richness, EPT richness, %EPT, mayfly taxa richness, and % mayflies all decreased with increasing conductivity and increasing % sand and fines (increasing sedimentation). In contrast, these same metrics all increased with increasing total habitat scores. The HBI and % Chironomidae metrics increased with increasing conductivity and % sand and fines. By comparison, these metrics decreased with increasing total habitat scores and sediment deposition scores. Correlations between the benthic metrics and selected physical and chemical variables indicate that the strongest and most significant associations were between biological condition and conductivity. West Virginia's aggregate bioassessment index (WV Stream Condition Index) and the mayfly taxa richness metric were the benthic metrics most strongly correlated to median conductivity ($r = -0.810$ and $r = -0.812$, respectively) (Green et al. 2000).

Table 5. Pearson correlation matrix for MBI and associated metrics and $\log_{10}$ transformed physicochemical parameters for all sites. Bolded values are **not** significant ($p<0.05$). RBP habitat metrics are based on scores (0-20).

	MBI	*TR*	*EPT*	*mHBI*	*m%EPT*	*%Ephem*	*%Chiro+Olig*	*%Clingers*	*TNI*
Catchment Area	**-0.24**	**-0.14**	**-0.15**	**0.32**	**-0.33**	**-0.10**	**0.38**	**-0.11**	**0.02**
pH	-0.68	-0.45	-0.60	0.68	-0.57	-0.60	0.68	-0.55	**-0.12**
Conductivity	-0.80	-0.62	-0.76	0.75	-0.66	-0.74	0.70	-0.63	**-0.29**
Total RBP Habitat Score	0.74	0.57	0.71	-0.76	0.68	0.61	-0.66	0.55	**0.31**
Bank Stability	**0.21**	**0.19**	**0.21**	**-0.20**	**0.26**	**0.13**	**-0.21**	**0.09**	**0.04**
Bank Vegetation	0.46	**0.38**	0.48	-0.50	0.42	**0.32**	-0.40	**0.36**	**0.22**
Channel Flow Status	**0.15**	**0.22**	**0.12**	**-0.11**	**0.17**	**0.04**	**-0.15**	**0.15**	**0.13**
Channel Alteration	0.43	**0.29**	0.43	-0.47	**0.37**	**0.38**	**-0.32**	**0.36**	**0.21**
Embeddedness	0.64	0.56	0.61	-0.65	0.53	0.52	-0.61	0.45	**0.27**
Epifaunal Substrate	0.54	**0.37**	0.54	-0.57	0.44	0.49	-0.43	0.45	**0.27**
Frequency of Riffles	**0.27**	**0.08**	**0.25**	**-0.32**	**0.27**	**0.28**	**-0.23**	**0.21**	**0.10**
Riparian Zone Width	0.45	**0.33**	0.44	-0.49	0.42	0.39	**-0.36**	**0.31**	**0.17**
Sediment Deposition	0.49	0.41	0.43	-0.49	0.42	**0.38**	-0.51	0.40	**0.30**
Velocity/Depth Regime	0.39	**0.38**	0.44	**-0.37**	**0.32**	**0.37**	**-0.27**	**0.25**	**0.35**
Canopy Cover	0.41	**0.34**	0.42	-0.44	**0.35**	**0.38**	**-0.27**	**0.31**	**0.29**

4.2.2.1 Distinguishing Land Use Disturbance with Specific Indicator Measurements

The stepwise DFA selected five indicator measurements (out of 8 biological and 20 habitat and physicochemical parameters) that best discriminated between the four land-use categories. These included three biological metrics (m%EPT [*F*=11.03], %Ephem [*F*= 7.35], mHBI [*F*=4.28]) and two physicochemical or habitat parameters (conductivity [*F*=12.62], and total habitat score [*F*=2.13]) that classified the *a priori* land-use categories with 87% efficiency. An internal jackknife test of the data also classified the sites with only a 15% misclassification rate. Overall, the five-variable discriminant model was highly significant (Wilk's λ= 0.083, *F*= 18.95, p<0.0001). Discriminant root scores are plotted in Figure 20. The five variables classified reference sites with 97% efficiency, mined sites with 75% efficiency, mined/residential sites with 83% efficiency, and residential sites with 78% efficiency. This information is useful for evaluating biological, chemical, and habitat data for gaining insight into the causes and sources of impairment. For example, mined sites are characterized by having high conductivity, low to moderate RBP habitat scores, few or no mayflies, moderate m%EPT, and a moderate to high mHBI value. In contrast, residential sites would be expected to have low to moderately elevated conductivity, low to moderate RBP habitat scores, high mHBI, low or high %Ephem, and moderate m%EPT.

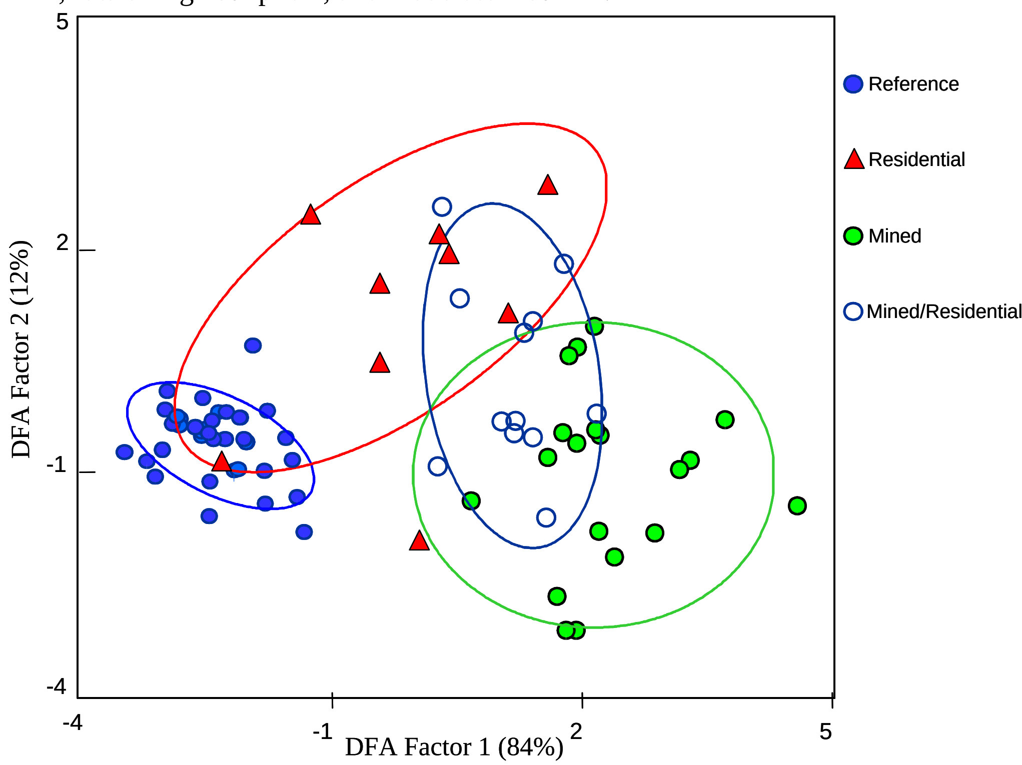

Figure 20. Discriminant function analysis plot of root scores from sites among the four land-use categories using a five-variable model (m%EPT, %Ephem, mHBI, conductivity, and total RBP habitat score).

4.2.2.2 *MBI and Metric Relationships to Conductivity and Habitat Quality*

The MBI showed a strong negative relationship to conductivity (Figure 21) (R^2=0.60, p<0.001, log-transformed data). Between the three disturbed landuse categories, a slight pattern was detected that might distinguish effects of land use on conductivity influences. Namely, the slope of the curves for residential and mined/residential were steeper than mined only sites. This suggests that factors other than conductivity are involved in MBI variability between land-use categories. The MBI responded positively to increasing habitat quality (R^2=0.54, p<0.001). Between the three disturbed land-use categories, no pattern was detected that might distinguish land-use-specific habitat influences (Figure 22).

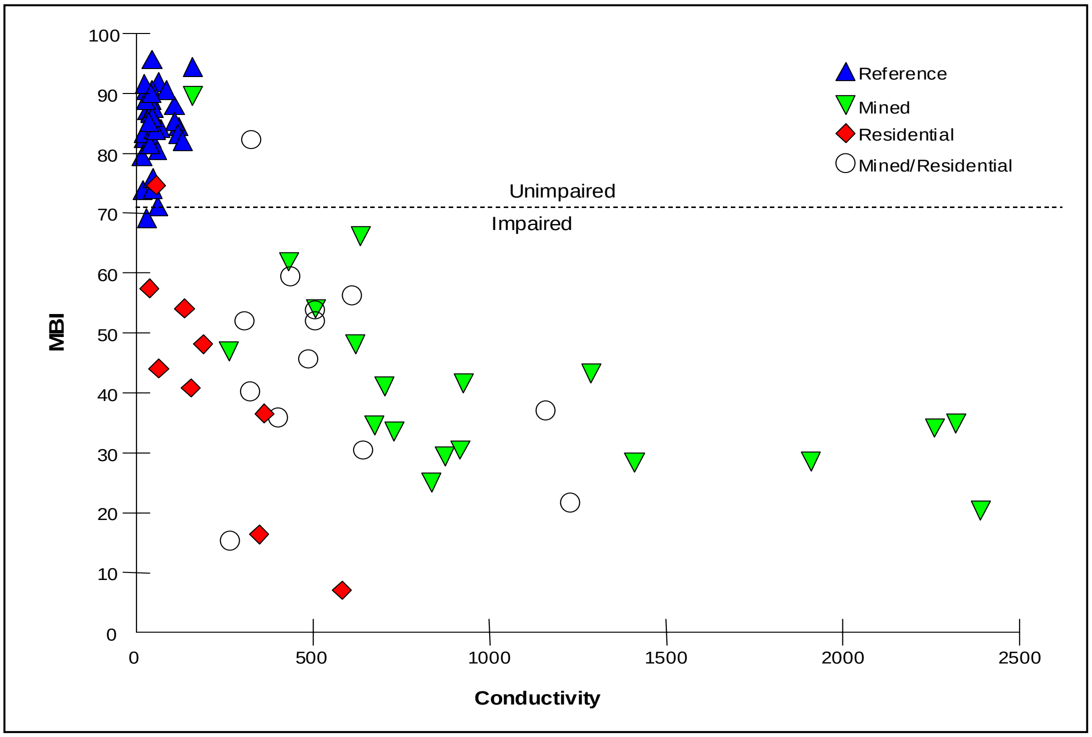

Figure 21. Scatterplot of MBI scores versus conductivity (µS/cm) by land-use category.

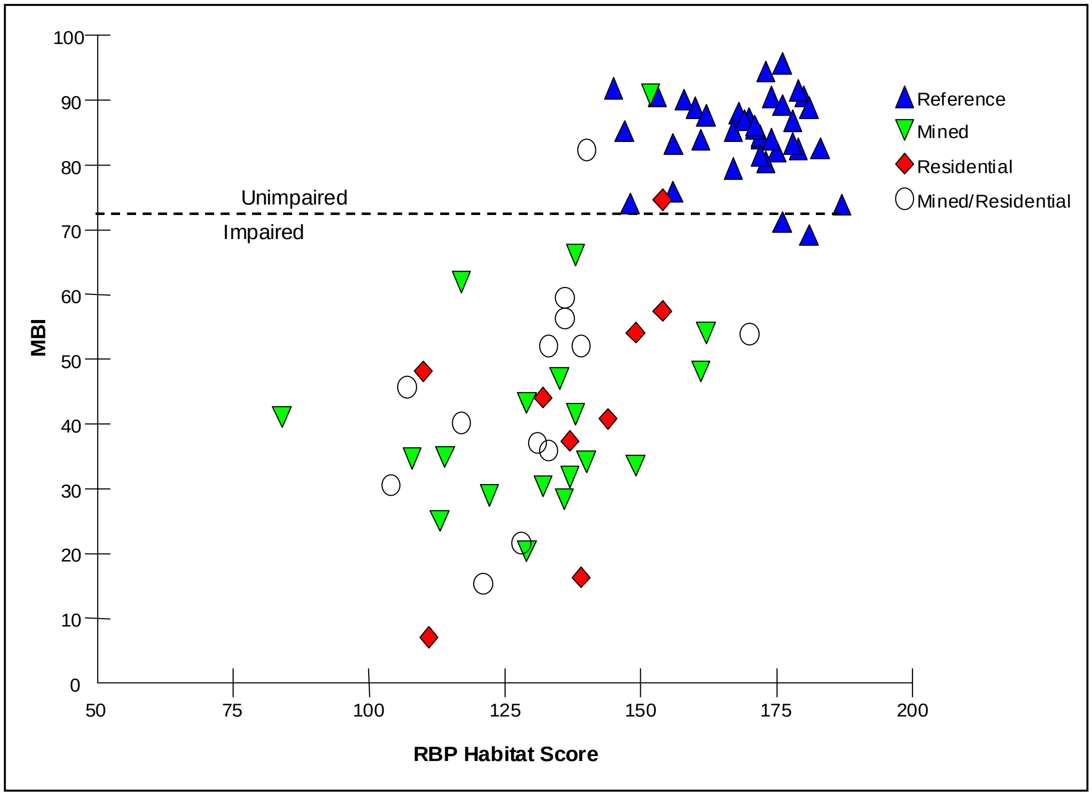

Figure 22. Scatterplot of MBI scores versus RBP habitat score by land-use category.

All seven of the metrics that make up the MBI responded predictably to conditions associated with both mining and residential disturbances (see Table 5). Metrics showed the highest significant relationships to conductivity and habitat quality. EPT richness declined considerably along an increasing conductivity gradient (Figure 23) at mined and mined/residential sites (R^2=0.55, $p<0.001$, log-transformed data). However, some residential sites with moderately low conductivity also displayed low EPT richness. This was likely attributed to nutrient loading or organic enrichment and habitat degradation. Also, EPT richness increased considerably along an increasing habitat quality gradient (R^2=0.47, $p<0.001$), but no clear patterns between the three disturbed categories were detected (Figure 24). EPT richness is probably the most sensitive indicator of stream condition throughout the U.S. (Resh and Jackson 1993, Barbour et al. 1999), and has been found to respond to mining impacts (Green et al. 2000, Howard et al. 2001, Garcia-Criado et al. 1999). In the present study, reference sites had significantly higher EPT richness, and these results indicate that many EPT taxa will disappear in the presence of both mining and residential impacts.

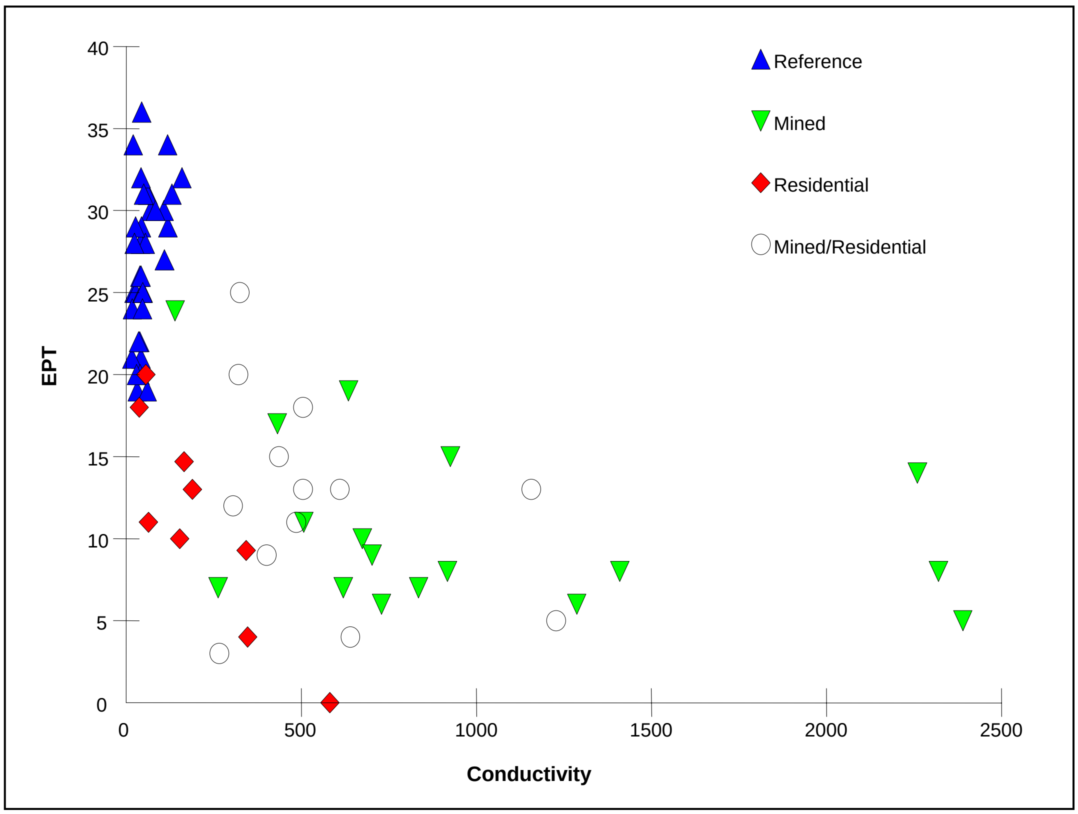

Figure 23. Scatter plot of EPT richness along conductivity gradient and among land-use categories.

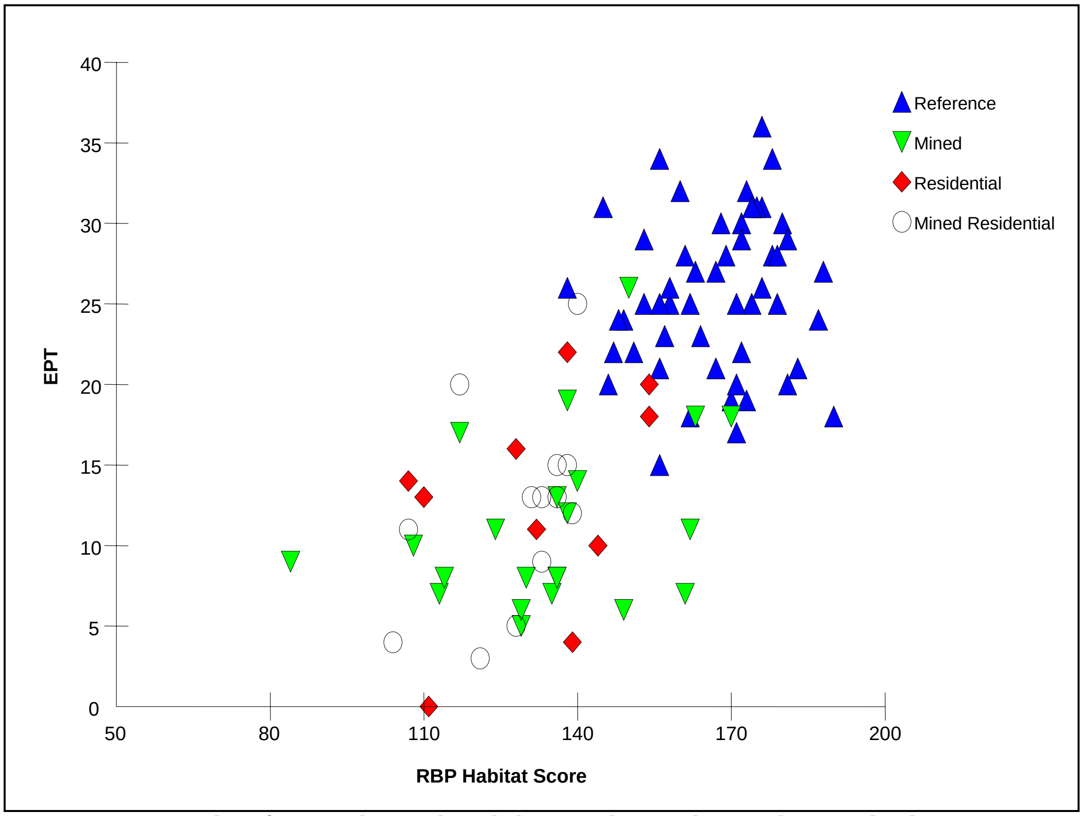

Figure 24. Scatter plot of EPT richness along habitat quality gradient and among land-use categories.

The EPT fauna can also be affected by other impacts such as timber harvesting. However, the duration of impairment can vary with the magnitude of the operation. For example, Stone and Wallace (1998) detected limited differences in macroinvertebrate community indices between two reference and clear-cut headwater streams in North Carolina. In fact, some increases in EPT richness were observed. While the authors noted significant increases in the NCBI (analogous to Kentucky's mHBI), those reported values would not indicate impairment in Kentucky. Increased richness and production of macroinvertebrates in the logged stream was in response to elevated light, temperature, and nutrients. They also noted changes in the food web or trophic structure of the communities. However, compared to mining, these disturbances are generally more benign and temporary (~5-10 years) and do not cause wholesale loss of sensitive taxa as was found in the present study. Moreover, only minor increases in conductivity may occur from logging. For example, one KDOW reference site in the Daniel Boone National Forest was heavily logged six years prior to sampling, but the conductivity was only 50 µS/cm, 32 EPT taxa were collected, and the MBI score was 83 (excellent).

Mayflies declined considerably along an increasing conductivity gradient (Figure 25) and especially at mined and mined/residential sites (R^2=0.60, p<0.001, log-transformed data). The sharp decline in the %Ephemeroptera metric indicated that these organisms are very sensitive to CMD. Moreover, residential sites produced similar harmful conditions for mayflies that may be linked to nutrient and organic loading. Mayfly abundance correlated less strongly with habitat (R^2=0.32, p<0.001), but the effect was significant (Figure 26).

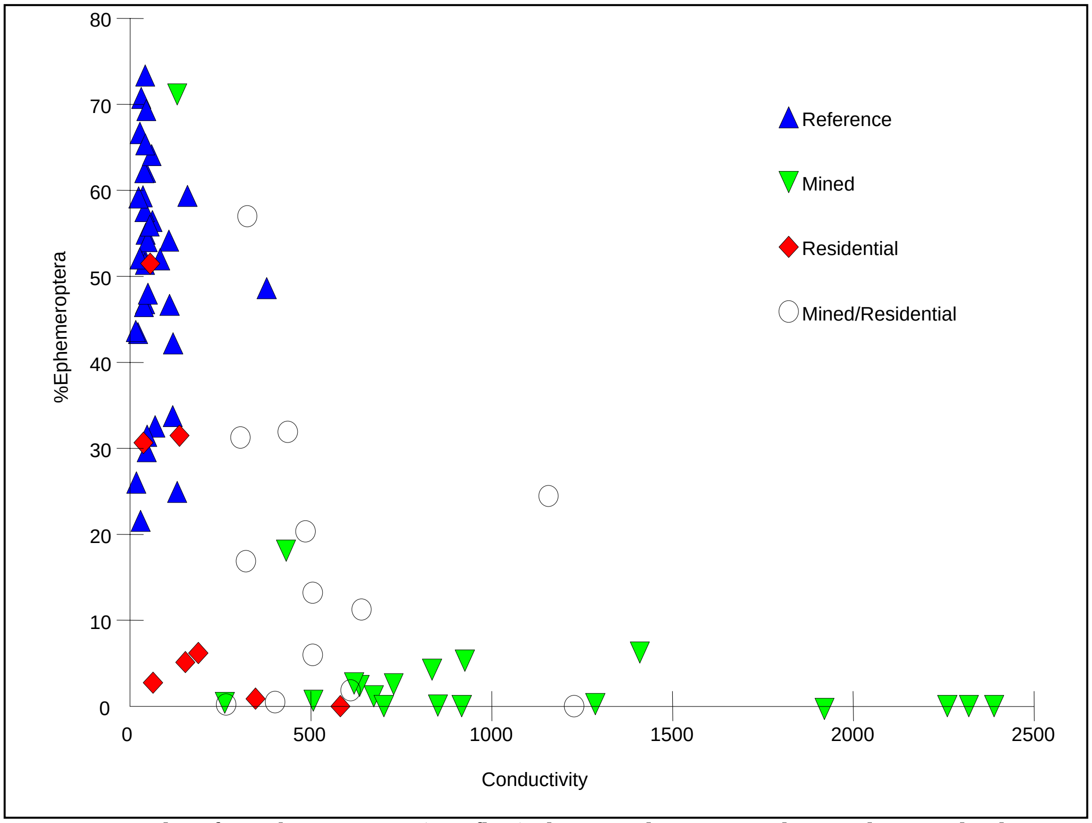

Figure 25. Scatterplot of %Ephemeroptera (mayflies) along conductivity gradient and among land-use categories.

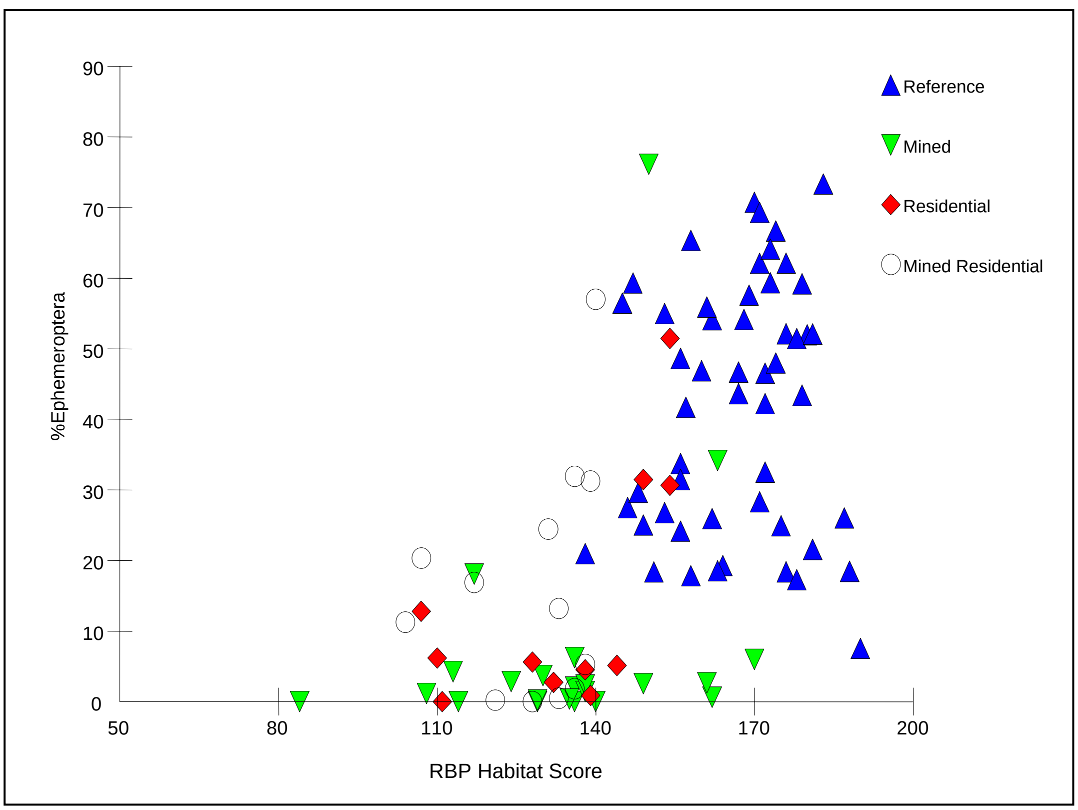

Figure 26. Scatterplot of %Ephemeroptera (mayflies) vs. RBP habitat scores and among land-use categories.

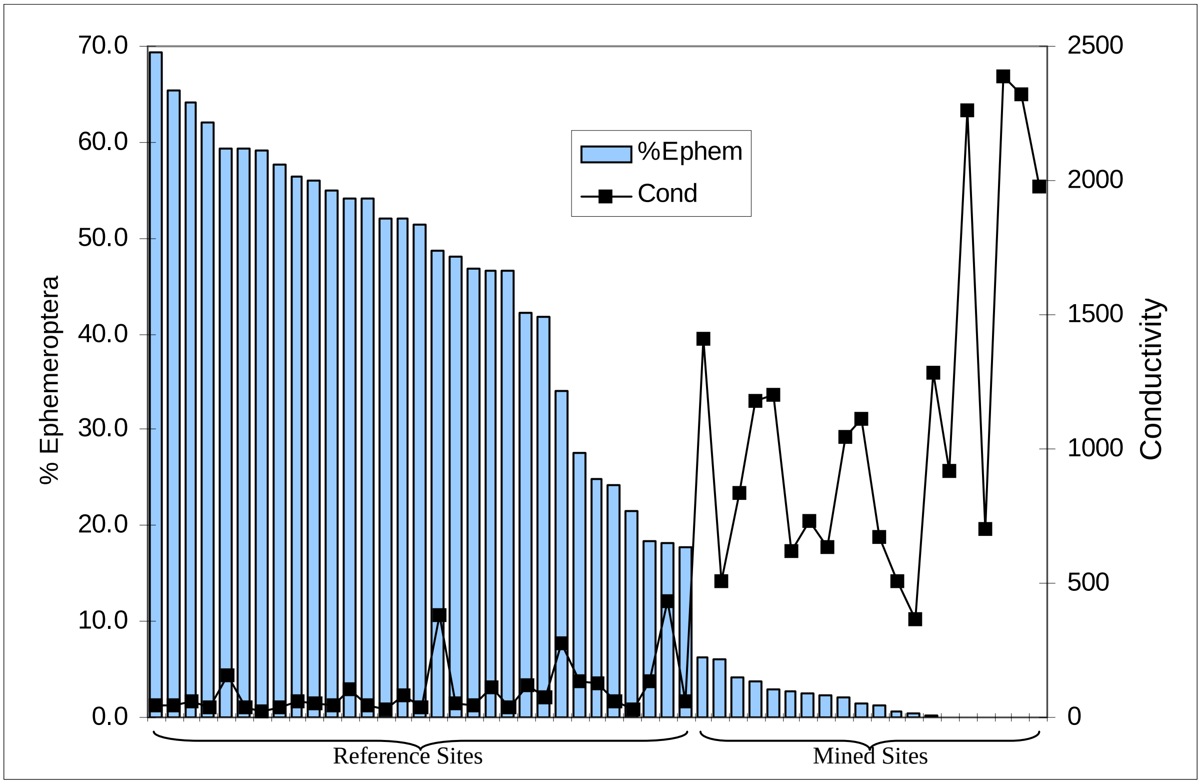

Figure 27. Bar/line chart showing %Ephemeroptera and conductivity (µS/cm) from reference and mined sites. Drastic reductions in mayflies occurred at sites with conductivities generally above 500 µS/cm.

The wholesale loss of mayflies at mined sites indicates that increased total dissolved solids (i.e., conductivity) from surface mining are harmful to these organisms. This relationship has been reported by Green et al. (2000) and Hartman et al. (2004) in West Virginia. Figure 27 emphasizes how elevated conductivity from surface mining impacts the relative abundance of ephemeropterans. Mayfly assemblages of usually ten or more species, and averaging nearly 50% of all organisms collected, dominate healthy headwater streams in the ECF. Figure 28 depicts decreases in mayfly richness among land-use categories. Clearly, mined sites had significantly lower richness compared to other categories. Interestingly, the boxplot inversely matches the boxplot of conductivity arranged by land-use categories (see Figure 14a). It is important to note that not all mayfly species are sensitive to high conductivity. Several facultative, warmwater mayflies (e.g., *Baetis*, *Isonychia*, *Caenis*, *Tricorythodes*) that are typically absent from reference sites can invade headwater habitats that have elevated conductivity, temperature, or nutrients (KDOW unpub. data). As with mine discharge, toxicity to some mayflies may occur from exposure to or ingestion of trace heavy metal compounds (Clements 1994) or purely from the rise in conductivity itself by interfering with osmoregulation (i.e., gill function and respiration). Further research on the mechanisms of mayfly toxicity is warranted.

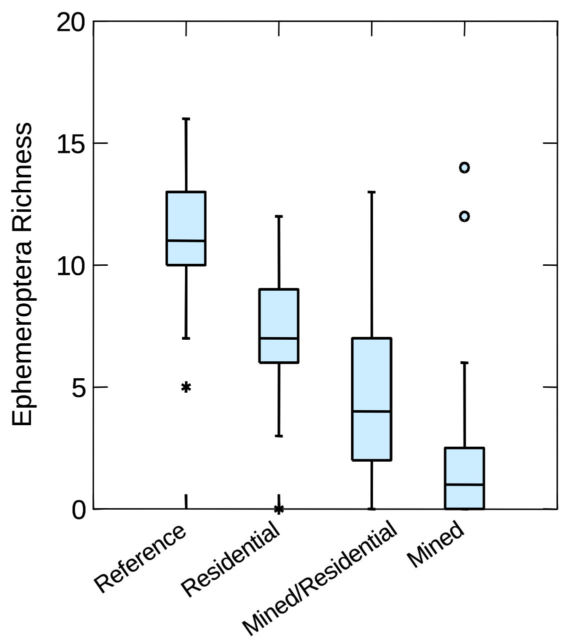

Figure 28. Boxplot of Ephemeroptera richness among land-use categories.

The loss of mayflies from some residential sites that had elevated nutrients or organic wastes could be due to observed filamentous bacterial infestations. This assumption is supported by a study by Lemly (1998, 2000) that showed 100% mortality of headwater mayfly taxa (e.g., *Epeorus*) when their bodies were more than 25% covered in *Sphaerotilus*. Stoneflies and caddisflies were also affected by *Sphaerotilus* infestations that resulted in poor growth and failure to reach maturity and emerge. Lemly also reported that even low to moderate increases in nitrogen and phosphorus can stimulate blooms of filamentous bacteria in normally nutrient-poor stream systems in the Appalachian Mountains. In the present study, many taxa were often found with bacterial growths on body surfaces. Although not all sites had corresponding water chemistry data, elevated nutrients (total phosphorus, nitrate, ammonia) and organic wastes (total organic carbon) were frequently found below residential and mined/residential areas with improper on-site wastewater treatment systems. General habitat degradation may also be partially responsible mayfly decline at residential sites.

The mHBI metric also showed a strong response to conductivity (R^2=0.56, p<0.001, log-transformed data) (Figure 28). This metric also responded strongly to habitat quality (R^2=0.58, p<0.001 log-transformed data) (Figure 29). The tightly clustered distribution of mHBI values further demonstrated the predictability of reference site expectations. Although this biotic index was originally formulated to detect organic pollution (Hilsenhoff 1988), these results showed that the metric responded well to inorganic chemical pollutants and habitat degradation associated with mining. This metric, or similar variants (e.g., North Carolina Biotic Index; Lenat 1993) has shown sensitivity to increased nutrient concentrations and habitat degradation (Pond et al. 2003) and insecticides (Wallace et al. 1996). Thus, assigned tolerance values indirectly integrate a wide variety of species response to stress.

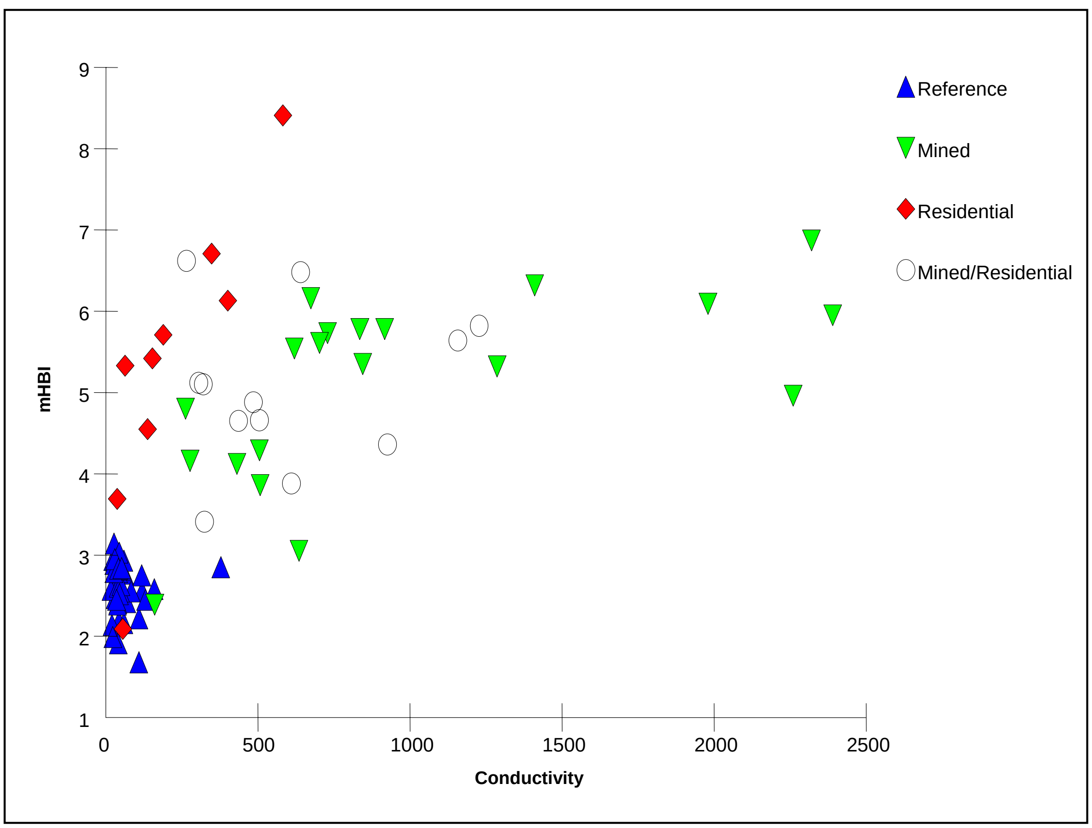

Figure 29. Scatterplot mHBI along conductivity gradient and among land-use categories.

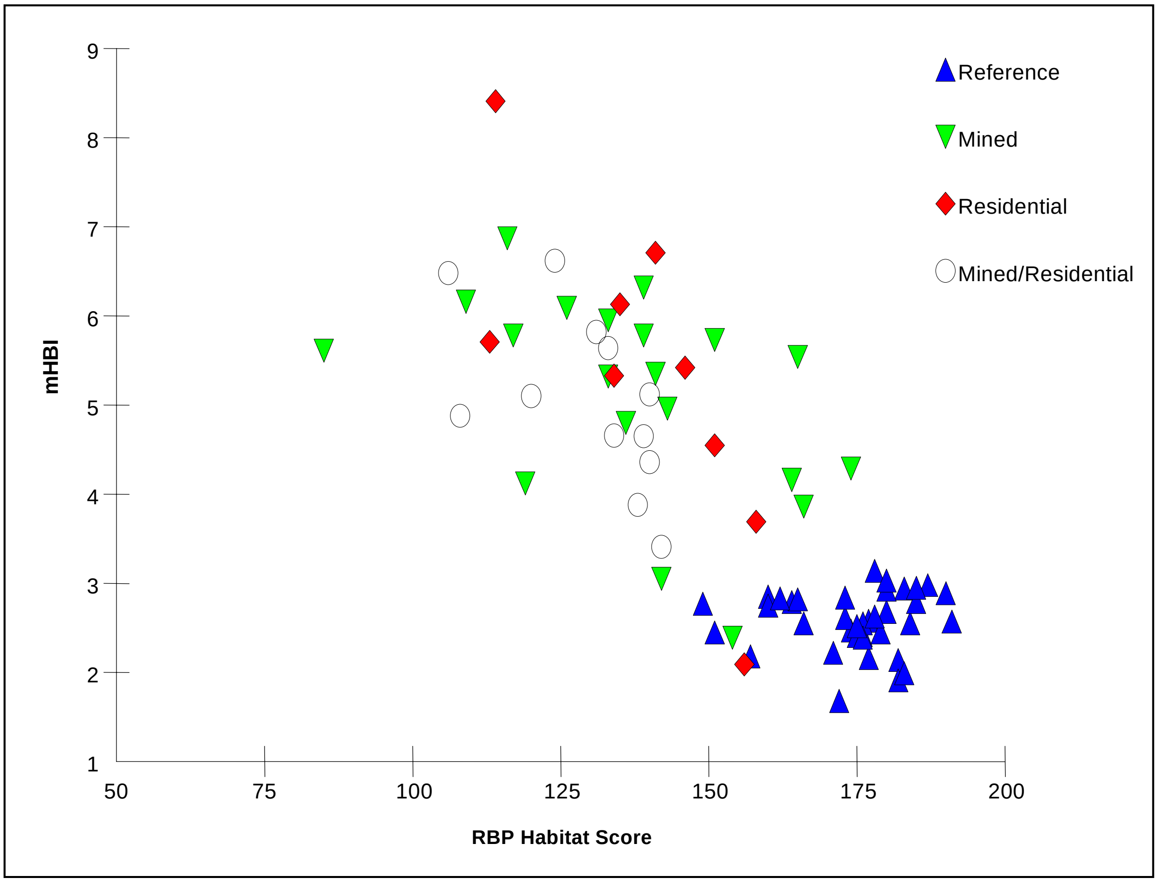

Figure 30. Scatterplot of mHBI vs. RBP habitat score and among land-use categories.

Although the DFA chose m%EPT as an indicator to distinguish land-use types, it had a lower correspondence to conductivity (R^2=0.37, p<0.001, log-transformed data) than most other metrics (Figure 31). It was found that some EPT taxa could tolerate elevated conductivity. For example, the nemourid stonefly *Amphinemura* may become fairly abundant in most degraded headwater streams as long as temperatures remain cool and detritus (i.e., food source) from riparian vegetation is available. The hydropsychid caddisflies *Hydropsyche betteni*, *Ceratopsyche bronta*, and *C. sparna* also represent EPT taxa that can tolerate elevated conductivity. This commonly used metric has been improved by excluding the hydropsychid caddisfly *Cheumatopsyche*, and it is possible that exclusion of other tolerant EPT taxa would strengthen this metric. The m%EPT metric showed a stronger relationship to habitat quality (R^2=0.46, p<0.001, log-transformed data) (Figure 32). This further demonstrates that this metric is good for diagnostic purposes when multiple stressors are responsible for impairment.

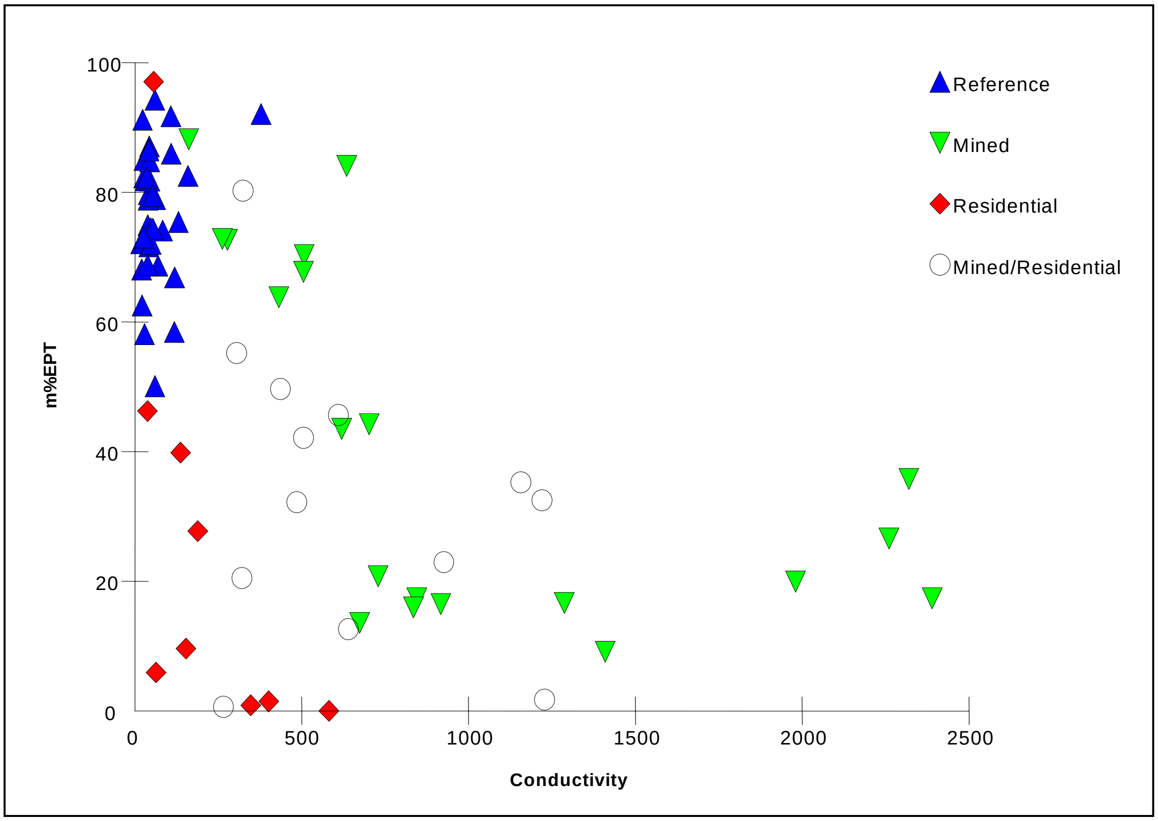

Figure 31. Scatterplot of m%EPT along conductivity gradient and among land-use categories.

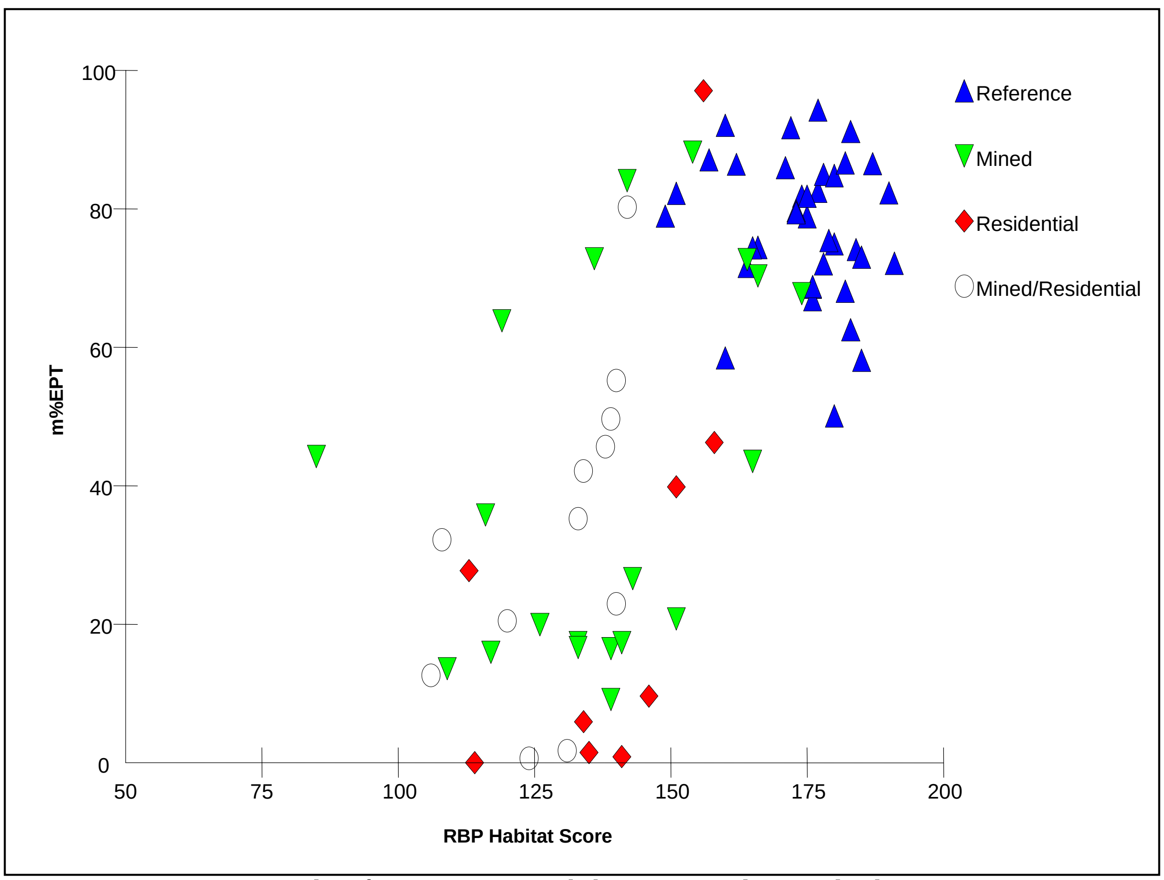

Figure 32. Scatterplot of m%EPT vs. RBP habitat score and among land-use categories.

4.3 Implications on Organism Health

Organism condition is not often evaluated in bioassessment protocols, especially with macroinvertebrates. However, some states (e.g., Ohio) evaluate fish health by calculating the percent of diseased individuals, fish with eroded fins, lesions, and tumors (DELT anomalies). Some workers have documented mouthpart deformities in chironomid larvae (e.g., Warwick 1988) due to heavy metal burdens, but this was not positively observed in the present study. Researchers at the University of Louisville (J. Jack, pers. comm.) have found specimens of dusky salamanders (*Desmognathus*) with missing or deformed limbs and polydactyly (extra "fingers") in streams with severe mine drainage problems. In the present study, invertebrates were often observed with bacterial and fungal infestations or coated in various mining-related precipitants. Although few data are available regarding the toxic or pathogenic effects of these substances, it is likely that their occurrence is detrimental to an organism's health and may interfere with growth or other life history requirements. Figure 33 shows a caddisfly with heavy mineral deposits on its integument, sclerites, and gills, while Figure 34 shows a caddisfly with a severe fungal infestation.

Figure 33. Hydropsychid caddisfly (*Hydropsyche betteni*) coated in iron and manganese precipitants. Circles highlight areas with Fe (thorax) or Mn (gills) deposition.

Figure 34. Philopotamid caddisfly (*Chimarra obscura*) with fungal infestation. The circle highlights fungal hyphae.

4.4 General Discussion on Mining and Residential Impacts

4.4.1 Mining

The MBI indicated that 95% of the mined sites were impaired. One mined site that was not impaired had conductivity and RBP habitat values similar to that of reference streams (~160 (μS/cm) and ~150, respectively). Subsequently, this site did not have hollowfills in its watershed. The most degraded mined site (as indicated by an MBI score of 19.8) had the highest conductivity (2350 μS/cm) and a moderately low habitat score (133). Unquestionably, the physicochemical effects of mining are important to biological communities. A study by U.S. EPA Region 3 in West Virginia (Green et al. 2000) reported that the increase in specific conductance and sulfate concentration was associated with a proportional decrease in the sensitive taxa in stream macroinvertebrate communities. Their study demonstrated that water chemistry explains the wide gradient in biological condition at hollowfilled sites. The hollowfilled sites that scored in the good and very good range were found to have better water quality, as indicated by lower median conductivity at these sites. The filled sites that scored in the fair, poor and very poor ranges had elevated median conductivity. A companion report by U.S. EPA (Fulk et al. 2003) also documented this occurrence. These results support findings from the present study indicating conductivity as a primary stressor of concern; however, sedimentation and general habitat degradation were also found to contribute to biological impairment.

Sediment pollution (i.e., siltation) is the number one stressor to aquatic life in Kentucky according to latest 305(b) reports. (KDOW 2004; KDOW 2002b). Sedimentation can impair aquatic life by reducing light penetration, smothering organisms and their habitats, and by introducing absorbed pollutants (e.g., metals and nutrients) (Lenat and Penrose 1981). In eastern Kentucky, KDOW also reported that resource extraction (i.e., surface mining) was the leading source of sedimentation. While there are regulations and technologies to control sedimentation from mine sites, the problem is still severe. Sediment ponds, while helping to control suspended solids during mining activities, may cause an initial sediment stress to the stream by pond construction itself. KDOW staff have found themselves “knee-deep” in sediment 50 m below a constructed pond. This sedimentation was apparently caused by the actual construction of the pond and inadequate BMP’s on pond outslopes. These sediment ponds can also increase water temperature and potentially alter food resources for downstream communities (high organic seston loading). While sestonic particles supply food for filter feeders (e.g., semi-tolerant hydropsychid and philopotamid caddisflies, and invasive Asiatic clams), the result is a wholesale alteration of the expected community structure. Furthermore, the modified thermal and flow regimes downstream of these ponds might interfere with invertebrate phenologies (e.g., adult emergence, egg hatching, growth and development) thereby altering important life history requirements.

There is also some evidence that surface mining in the ECF increases nutrient loading to receiving streams. Nitrate concentrations may reach several milligrams per liter below hollowfills, even after 10 or more years following reclamation (KDOW and J. Jack, U of L, unpub. data). However, a study in West Virginia (U.S. EPA 2002b) showed only some of the hollowfilled sites had elevated nitrate, while many had concentrations similar to background conditions. Excessive nutrient additions to these normally nutrient poor stream systems alters community structure and may cause nuisance blooms of algae or filamentous bacteria, thereby directly affecting macroinvertebrate assemblages.

The long-term impacts to these headwater streams cause problems for re-colonization by indigenous macroinvertebrate communities. In many mining situations much of the most intense disturbance occurs at the stream origin and progresses downstream, which means that few or no organisms may be available to re-colonize the affected streams after elimination of the organisms by physical disturbance or chemical toxicity. Aerial dispersal from adjacent tributaries (if not impacted) would be the only source of colonization. U.S. EPA (2002a) estimated dissolved solids loading may last in excess of 25 years. However, geologists at DEP surmise that high chemical loading of dissolved solids may persist for centuries, as crushed overburden weathers in hollowfills. This is further complicated by the fact that there are no valid treatment technologies available for this type of discharge from current mining practices. In addition, reclaimed mine lands in Kentucky are mostly converted to grasslands rather than the pre-mining forested landscape. This could also have a negative impact on stream functions and invertebrate community structure (U.S. EPA 2002a). While it is important to restore headwater stream habitat following mining or other major land-moving activity (e.g., highway construction), stream communities may continue to be hampered by chemical pollution. For example, Figure 35 shows a re-aligned stream channel after mining. Downstream (Figure 36), a relatively undisturbed, forested reach was still highly impaired (MBI score in the "Very Poor" range). Here, substrates were armored with heavy mineral deposits, and the conductivity was greater than 2000 μS/cm.

Figure 35. View of a realigned headwater stream in Floyd Co. following surface mining.

Figure 36. Downstream view of the stream in Figure 35 showing more natural stream habitat. However, the conductivity was 2350 μS/cm and MBI score was 19.8 (Very Poor).

4.4.2 Residential

The MBI revealed that close to 90% of the streams with residential land use were impaired. One residential site that was not impaired had low conductivity (56 μS/cm) and good instream habitat (RBP Habitat Score=156). Impaired sites also had low conductivity, but were most often habitat limited and showed signs of elevated nutrients or organic wastes. The worst residential site (as indicated by its MBI score of 7.1) had elevated conductivity (~500 μS/cm), degraded habitat (RBP Habitat Score=114), and highly elevated ammonia, nitrate, and total phosphorus concentrations (0.653, 0.913, and 0.231 mg/L, respectively). It also had the highest number of residences above the sample point. In eastern Kentucky, much of the human settlement occurs along relatively small streams because of topographic limitations. Even low-density housing can cause impacts to streams in narrow valleys or hollows. Because of the close proximity to stream channels, pollutant loading, riparian forest destruction and stream channelization are common. The data presented herein showed that residential sites, although somewhat variable, had significantly impacted macroinvertebrate communities.

Some of the residential and mined/residential sites used in this report had elevated ammonia, nitrate, and total phosphorus levels (KDOW unpub. data). Nutrient enrichment in normally nutrient poor streams can stimulate nuisance algal growth and filamentous bacteria such as *Sphaerotilus* that can cause deleterious effects to resident macroinvertebrate communities (Lemly 1998). Nutrient enrichment comes from straight-pipe sewage and failing septic systems, as well as storm water runoff from gardens and lawns. Kentucky ranks high in the U.S. in the number of inadequate septic systems, much of it concentrated in eastern Kentucky.

Besides nutrient additions from residential development, discharge of household chemicals and detergents directly into streams can cause harm to aquatic organisms. KDOW biologists have frequently observed soapy or oily discharges from graywater straightpipes. Other potential pollutants from homes include oil, grease and other petroleum hydrocarbons, heavy metals, litter and debris, animal wastes, solvents, paint and masonry wastes, detergents and other cleaning solutions, and pesticides and fertilizers. Although little or no data exist on the effects of these pollutants in the ECF, organisms living in small headwater streams with minimal dilution capacity are undoubtedly exposed to these chemical substances.

Road density was generally higher in residential areas, with the potential to cause greater sedimentation in nearby streams. In most of the residential sites investigated by KDOW, stream channels were burdened with excessive silt and sediment loads. Stream channels flowing through residential areas were also likely to have once been re-aligned or channelized (see Figure 9). Finally, the loss of streamside forests in residential areas elevates the stream's temperature and consequently alters invertebrate life history cues.

4.4.2 Mined/Residential

The MBI revealed that roughly 93% of the mined/residential sites were impaired. These sites were expected to show the greatest impairment, receiving multiple chemical and physical stresses associated with both land uses. Although mean MBI scores were lower for this land-use category, the difference was not significant. In West Virginia, Fulk et al. (2003) found that filled/residential sites scored the lowest macroinvertebrate index score compared to filled or unmined sites. In the present study, the single mined/residential site that was not impaired had slightly elevated conductivity (324 μS/cm) and moderate instream habitat (RBP Habitat Score=140). The worst mined/residential site (as indicated by its MBI score of 15.4) had slightly elevated conductivity (~265 μS/cm), and degraded habitat (RBP Habitat Score=120), and slightly elevated nitrate and total phosphorus (0.795 and 0.02 mg/L, respectively).

Overall, mined/residential sites had elevated conductivity, increased nutrient concentrations, physical habitat degradation, and more sediment than reference sites. The combination of all of the potential stressors listed for mined and residential sites apply to mined/residential sites, and it was no surprise that this land-use type caused widespread impairment.

5.0 Conclusions

Results from this investigation revealed that both surface mining and residential land uses have negative effects on macroinvertebrates in headwater streams in the ECF. Historical impacts (e.g., logging, agriculture) from the early 1900's cannot be blamed since even high quality reference streams were exposed to those same historical impacts. Timber harvesting can undoubtedly impact headwater stream communities, but these impacts may not be as enduring as mining operations or residential occupation. Statistically significant departures from reference conditions were noted for several physical and biological parameters commonly used in water quality assessments. Both physical (e.g., sedimentation, loss of riparian vegetation and canopy cover, and instream habitat quality) and chemical (e.g., increases in conductivity, pH, and nutrients or organic wastes) factors appear to operate separately and in combination to cause biological impairment by altering the predicted, natural macroinvertebrate community.

Dissolved solids emanating from hollowfills are a primary cause of biological impairment because of their severe impact to mayflies (a key component of headwater stream communities) and other sensitive taxa. Although some land-use specific responses were found with physical, chemical, and biological data, few significant differences were detected between mined and residential watersheds, indicating that both land uses can equally impair aquatic life. Both mining and residential impacts are unquestionably long term, and although certain impacts are avoidable, they are not likely to be eliminated by current regulatory efforts.

Residential developments along headwater stream corridors cause considerable long-term physical and biological impacts. Moderate to heavy housing densities in areas with inadequate wastewater treatment results in discharges of raw sewage and a variety of household chemicals. Although minimal chemical data exists for straight-pipe discharges, data presented here showed that residential land use led to significant decreases in MBI scores relative to the reference condition.

When mining and residential developments both occur in a watershed, aquatic communities are faced with a combination of stressors (nutrient and organic enrichment, elevated dissolved solids) as well as physical habitat degradation. Although this study did not detect patterns that might distinguish mined/residential from mined or residential, the synergistic effects of both land uses will continue to impair waterbodies in the ECF.

Finally, it is important to acknowledge that headwater streams serve as “capillaries,” functioning to convey clean water and food resources to downstream communities and human uses. Healthy headwater streams in the ECF support diverse assemblages of sensitive macroinvertebrates and help to define “natural” stream ecosystems. While most of the data used in this study came from perennial streams, many sites were intermittent. Several studies have indicated that there is little, if any, difference in macroinvertebrate assemblages between intermittent and perennial reaches (Delucchi 1988, Feminella 1996, Green et al. 2000). Disruptions in the ecological processes of first- and second-order streams impact not only aquatic life and water quality within the stream, but also the functions that are contributed to downstream aquatic systems in the form of nutrient cycling, food web dynamics, and species diversity (Cummins 1980, Merritt et al. 1984). Doppelt et al. (1993) stressed the value of headwater streams by stating that: “Even where inaccessible to fish, these small streams provide high levels of water quality and quantity, sediment control, nutrients and wood debris for downstream reaches of the watershed. Intermittent and ephemeral headwater streams are, therefore, often largely responsible for maintaining the quality of downstream riverine processes and habitat for considerable distances.” Thus, curtailing environmental disturbance or restoring impacted headwater streams should be the first step in improving downstream functions and uses.

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Appendix A. List of headwater streams sampled for macroinvertebrates in the Eastern Coalfield Region.

StationID	Stream Name	Category	Basin	Sub-Basin	Collection Year	Order	Area (sq.mi.)	Ecoregion	County	Topo Name
1005013	VENTERS BR.	Mined	BIG SANDY	TUG FORK	2004	1	0.49	69	MARTIN	INEZ
1005014	MUDLICK BR. UT	Mined	BIG SANDY	TUG FORK	2004	1	0.36	69	MARTIN	INEZ
1005015	LICK BR.	Mined	BIG SANDY	TUG FORK	2004	2	0.86	69	MARTIN	INEZ
1007005	HOBBS FK.	Reference	BIG SANDY	TUG FORK	2001	2	1.15	69	MARTIN	VARNEY
1007006	HOBBS FK. UT	Reference	BIG SANDY	TUG FORK	2001	1	0.18	69	MARTIN	VARNEY
1007012	PANTHER FK.	Mined	BIG SANDY	TUG FORK	2004	2	1.69	69	MARTIN	THOMAS
1007013	RIGHT FK. PANTHER FK.	Mined	BIG SANDY	TUG FORK	2004	1	0.47	69	MARTIN	THOMAS
1007014	WHITECABIN BR. UT	Mined	BIG SANDY	TUG FORK	2004	1	0.54	69	MARTIN	INEZ
1011001	LOWER ELK CR.	Mined/Residential	BIG SANDY	TUG FORK	2002	2	1.46	69	PIKE	MAJESTIC
1017003	STRATTON BR.	Mined	BIG SANDY	LEVISA FORK	2004	1	0.74	69	FLOYD	LANCER
1022001	SALISBURY BR.	Residential	BIG SANDY	LEVISA FORK	2002	1	1.65	69	KNOTT	WAYLAND
1022002	SIZEMORE BR.	Residential	BIG SANDY	LEVISA FORK	2002	1	1.65	69	FLOYD	WAYLAND
1022008	CALEB FK.	Residential	BIG SANDY	LEVISA FORK	2002	2	1.78	69	FLOYD	WHEELWRIGHT
1022009	OTTER CR.	Residential	BIG SANDY	LEVISA FORK	2002	2	3.3	69	FLOYD	WHEELWRIGHT
1022010	ARKANSAS CR.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	3	2.8	69	FLOYD	HAROLD
1022011	BUCK BR.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	2	2.1	69	FLOYD	MARTIN
1022013	STEPHENS BR.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	2	2.2	69	FLOYD	MARTIN
1022014	JOHNS BR.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	1	0.8	69	FLOYD	MARTIN
1022016	WILSON CR.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	2	3.1	69	FLOYD	MARTIN
1022017	GOOSE CR.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	1	1.3	69	FLOYD	WAYLAND
1022021	STEELE CR.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	2	3.4	69	FLOYD	WAYLAND
1022024	BILL D BR.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	3	3.7	69	KNOTT	KITE
1022026	ARNOLD FK.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	2	3.5	69	KNOTT	KITE
1022029	SIMPSON BR.	Mined/Residential	BIG SANDY	LEVISA FORK	2002	2	1.9	69	FLOYD	MCDOWELL
1031001	WOLFPEN BR.	Mined	BIG SANDY	LEVISA FORK	2002	2	0.1	69	PIKE	ELKHORN CITY
1032001	TOMS BR.	Reference	BIG SANDY	LEVISA FORK	2001	1	0.95	69	PIKE	HELLIER
1032002	LOWER PIGEON BR.	Reference	BIG SANDY	LEVISA FORK	2001	1	0.89	69	PIKE	CLINTWOOD
1032003	UPPER PIGEON BR.	Mined	BIG SANDY	LEVISA FORK	2002	2	2	69	PIKE	JENKINS EAST
4036017	STEER FK.	Reference	KENTUCKY	KENTUCKY	2001	2	3	70	JACKSON	MCKEE
4036022	HUGHES FK.	Reference	KENTUCKY	KENTUCKY	2001	1	1.35	70	JACKSON	MCKEE
4042016	MIDDLE FK. RED RIVER	Residential	KENTUCKY	RED	2002	2	1.8	70	WOLFE	ZACHARIAH
4050007	FUGATE FK.	Mined/Residential	KENTUCKY	N. FORK KENTUCKY	2000	2	2.6	69	BREATHITT	NOBLE
4050008	JENNY FK.	Mined	KENTUCKY	N. FORK KENTUCKY	2000	1	0.45	69	BREATHITT	NOBLE
4050009	BEAR BR.	Mined	KENTUCKY	N. FORK KENTUCKY	2000	2	1.54	69	BREATHITT	NOBLE
4050010	CLEMONS FK.	Reference	KENTUCKY	N. FORK KENTUCKY	2000	2	0.8	69	BREATHITT	NOBLE
4050011	FALLING ROCK BR.	Reference	KENTUCKY	N. FORK KENTUCKY	2000	1	0.41	69	BREATHITT	NOBLE
4050012	JOHN CARPENTER FK.	Reference	KENTUCKY	N. FORK KENTUCKY	2000	1	0.58	69	BREATHITT	NOBLE
4050013	SHELLY ROCK FK.	Reference	KENTUCKY	N. FORK KENTUCKY	2000	1	0.55	69	BREATHITT	NOBLE
4050014	MILLSEAT BR.	Reference	KENTUCKY	N. FORK KENTUCKY	2000	2	0.58	69	BREATHITT	NOBLE
4050015	LITTLE MILLSEAT BR.	Reference	KENTUCKY	N. FORK KENTUCKY	2000	2	0.82	69	BREATHITT	NOBLE
4050016	LICK BR.	Mined	KENTUCKY	N. FORK KENTUCKY	2000	2	2.81	69	PERRY	NOBLE
4050017	WILLIAMS BR.	Mined/Residential	KENTUCKY	N. FORK KENTUCKY	2000	2	1.08	69	PERRY	NOBLE
4050018	CANEY CR.	Residential	KENTUCKY	N. FORK KENTUCKY	2000	2	2.5	69	BREATHITT	HADDIX
4050019	ROARING FK.	Reference	KENTUCKY	N. FORK KENTUCKY	2003	1	0.38	69	BREATHITT	NOBLE
4052017	LITTLE DOUBLE CR.	Reference	KENTUCKY	S. FORK KENTUCKY	2000	2	1.5	69	CLAY	BIG CREEK
4052018	RIGHT FK. BIG DOUBLE CR.	Reference	KENTUCKY	S. FORK KENTUCKY	2000	2	1.46	69	CLAY	CREEKVILLE
4052019	LEFT FK. BIG DOUBLE CR.	Reference	KENTUCKY	S. FORK KENTUCKY	2000	2	0.6	69	CLAY	CREEKVILLE
4052020	RIGHT FK. ELISHA CR.	Reference	KENTUCKY	S. FORK KENTUCKY	2000	2	2.35	69	LESLIE	CREEKVILLE
4052021	BIG MIDDLE FK. ELISHA CR.	Reference	KENTUCKY	S. FORK KENTUCKY	2000	1	0.82	69	CLAY	CREEKVILLE
4052022	LEFT FK. ELISHA CR.	Reference	KENTUCKY	S. FORK KENTUCKY	2000	2	2.47	69	LESLIE	CREEKVILLE
4052023	RIGHT FK. BIG DOUBLE CR.	Reference	KENTUCKY	S. FORK KENTUCKY	2000	2	1.53	69	CLAY	CREEKVILLE
4052024	RED BIRD CR.	Mined/Residential	KENTUCKY	S. FORK KENTUCKY	2000	2	1.4	69	BELL	BEVERLY
4052026	LAWSON CR.	Mined/Residential	KENTUCKY	S. FORK KENTUCKY	2000	2	1.48	69	BELL	BEVERLY
4052027	SPRUCE BR.	Mined	KENTUCKY	S. FORK KENTUCKY	2000	2	0.95	69	CLAY	BEVERLY
4052028	GILBERTS LITTLE CR.	Residential	KENTUCKY	S. FORK KENTUCKY	2000	2	1.47	69	CLAY	CREEKVILLE
4052029	ARNETTS FK.	Residential	KENTUCKY	S. FORK KENTUCKY	2000	2	1.42	69	CLAY	CREEKVILLE
4052030	SUGAR CR.	Reference	KENTUCKY	S. FORK KENTUCKY	2000	2	3.05	69	LESLIE	CREEKVILLE
4054005	CAWOOD BR. UT	Reference	KENTUCKY	M. FORK KENTUCKY	2001	1	0.8	69	LESLIE	BLEDSOE

Appendix A. List of headwater streams sampled for macroinvertebrates in the Eastern Coalfield Region.

4054007	LEFT FK. CAMP CR.	Mined	KENTUCKY	M. FORK KENTUCKY	2001	1	0.93	69	LESLIE	CUTSHIN
4054008	CAMP CR.	Mined/Residential	KENTUCKY	M. FORK KENTUCKY	2001	2	2.7	69	LESLIE	CUTSHIN
4054009	BILL BR.	Reference	KENTUCKY	M. FORK KENTUCKY	2001	2	2.3	69	LESLIE	BLEDSOE
4054010	HONEY BR.	Reference	KENTUCKY	M. FORK KENTUCKY	2001	2	0.82	69	LESLIE	CUTSHIN
4055002	LINE FK. UT	Reference	KENTUCKY	N. FORK KENTUCKY	2000	1	0.22	69	LETCHER	ROXANA
4059012	LEFT FK. MILLSTONE CR.	Mined	KENTUCKY	N. FORK KENTUCKY	2003	1	0.81	69	LETCHER	MAYKING
4059013	RIGHT FK. MILLSTONE CR.	Mined	KENTUCKY	N. FORK KENTUCKY	2003	1	1.2	69	LETCHER	MAYKING
5037002	BOTTS FK.	Reference	LICKING	UPPER LICKING	2002	3	3.38	70	MENIFEE	SCRANTON
5037004	WELCH FK.	Reference	LICKING	UPPER LICKING	2002	2	1.5	70	MENIFEE	SCRANTON
6012003	NICHOLS FK.	Reference	LITTLE SANDY	LITTLE FORK LITTLE SANDY	2002	2	0.65	70	ELLIOTT	ISONVILLE
6012004	MEADOW BR.	Reference	LITTLE SANDY	LITTLE FORK LITTLE SANDY	2002	2	0.93	70	ELLIOTT	MAZIE
2024705	MILL CR.	Reference	CUMBERLAND	ROCKCASTLE	2001	2	2.6	68	JACKSON	MCKEE
2006027	HATCHELL BR.	Residential	CUMBERLAND	CUMBERLAND	2000	1	0.35	68	MCCREARY	CUMBERLAND FALLS
2006030	JACKIE BR.	Reference	CUMBERLAND	CUMBERLAND	2000	2	1.14	68	WHITLEY	SAWYER
2006031	CANE CR.	Reference	CUMBERLAND	CUMBERLAND	2000	1	0.65	68	WHITLEY	CUMBERLAND FALLS
2008017	ROCK CR. UT	Reference	CUMBERLAND	BIG S. FORK CUMBERLAND	2000	1	0.82	68	MCCREARY	BELL FARM
2008018	WATTS BR.	Reference	CUMBERLAND	BIG S. FORK CUMBERLAND	2000	2	2.2	68	MCCREARY	BELL FARM
2008019	PUNCHEONCAMP BR.	Reference	CUMBERLAND	BIG S. FORK CUMBERLAND	2000	2	1.7	68	MCCREARY	BELL FARM
2008020	ROCK CR. UT	Reference	CUMBERLAND	BIG S. FORK CUMBERLAND	2000	2	0.63	68	MCCREARY	BELL FARM
2008021	ROCK CR. UT	Reference	CUMBERLAND	BIG S. FORK CUMBERLAND	2000	1	0.37	68	MCCREARY	BELL FARM
2008022	ROCK CR. UT	Reference	CUMBERLAND	BIG S. FORK CUMBERLAND	2000	2	0.89	68	MCCREARY	BARTHELL
2014004	JENNEYS BR. UT	Residential	CUMBERLAND	CUMBERLAND	2000	1	0.66	68	MCCREARY	WHITLEY CITY
2041003	BROWNIES CR.	Reference	CUMBERLAND	CUMBERLAND	2000	2	2.3	69	HARLAN	EWING
2042002	EWING CR.	Mined/Residential	CUMBERLAND	CUMBERLAND	2000	2	3.06	69	HARLAN	HARLAN
2042003	WATTS CR.	Reference	CUMBERLAND	CUMBERLAND	2001	2	0.85	69	HARLAN	WALLINS CREEK
2046004	PRESLEY HOUSE BR.	Reference	CUMBERLAND	POOR FORK CUMBERLAND	2000	2	0.9	69	LETCHER	WHITESBURG
2046005	FRANKS CR.	Mined/Residential	CUMBERLAND	POOR FORK CUMBERLAND	2000	2	1.36	69	LETCHER	WHITESBURG