Evaluation of Current Measures of Aquatic Biological Integrity in the Central Appalachian Coalfields: Efficacy and Implications

Sara Sweeten
Department of Fisheries and Wildlife Conservation, Virginia Tech, Blacksburg, Virginia

Jerry Sweeten
Department of Biology, Manchester College, North Manchester, Indiana

John Craynon
Virginia Center for Coal and Energy Research, Virginia Tech, Blacksburg, Virginia

W. Mark Ford
U.S. Geological Survey, Virginia Cooperative Fish and Wildlife Research Unit, Department of Fisheries and Wildlife Conservation, Virginia Tech, Blacksburg, Virginia

Stephen Schoenholtz
Department of Forest Resources and Environmental Conservation, Virginia Tech, Blacksburg, Virginia

ABSTRACT

Within the central Appalachian Coalfields, efforts to minimize and mitigate impacts from surface coal mining have been complicated by incongruent restoration goals and poorly validated post-mine monitoring protocols that are unable to quantify ecosystem change relative to system recovery. At present, assessment techniques range from simple single species examinations to complex multi-taxa/life history decision matrices believed capable of documenting cumulative and additive cross-system impacts. Additionally, some effort has been placed in using abiotic measures or narrow biotic components as system surrogates to comment on stream health of mined streams. These metrics may not accurately indicate biological change particularly when extended out to predict impacts and assess recovery. Benthic macroinvertebrates are valuable indicator species. However, aquatic ecosystem function may not always be able to be inferred from macroinvertebrate assemblages alone, and full assessment of stream health may require use of a suite of taxa such as periphyton, macroinvertebrates, fish, and/or salamanders.

INTRODUCTION

Central Appalachia has high levels of terrestrial and aquatic biodiversity and species richness, with many endemic species (Morse et al. 1993). The New River catchment alone is estimated to have 23 endemic plant and animal species (Hocutt et al. 1978). The aquatic systems in this region contain some of the highest levels of biodiversity in North America, including ten percent of global salamander and freshwater mussel diversity (Stein et al. 2000; Bernhardt and Palmer 2011). However, many of these species are listed as threatened or endangered (Bernhardt and Palmer 2011). For example, 46 species of freshwater mussels once inhabited the Powell River in Virginia and Tennessee (Ortmann 1918). Currently 36 species, seven of which are listed as federally endangered, remain (Ahlstedt et al. 2005). In the United States, the biggest threat to biodiversity is loss/degradation of habitat (Wilcove et al. 1998).

Most stream habitats in the eastern half of the United States, including those in central Appalachia, have been significantly altered by past or current anthropogenic disturbances (Karr 1991). From the time of the European settlement of Central Appalachia in the 1700s through the mid-1900s, Central Appalachian streams were commonly used as unchecked drainages for residential wastewater, mining operation byproducts, and industrial discharges. The arrival of the railroads in the late 1800s brought large scale clearing and degradation of the land from exploitative practices such as commercial logging and coal mining that essentially impacted all of central Appalachian forests (Hibbart 1990; Nowacki and Abrams 1997).

In the 1960s and 1970s, researchers, policy makers, and industry became more aware of the degraded nature of not only central Appalachian streams but streams nationwide, which, along with a shift in society's view of environmental degradation, brought widespread, sweeping legislation and policy reform. The Federal legislation most influential to current coal mining practices are the Clean Water Act (CWA) of 1972, and the Surface Mining Control and Reclamation Act (SMCRA) of 1977 (Craynon et al. 2012). The CWA sets Federal regulatory standards for the identification, reporting, monitoring, and mitigation of degraded waters including those impacted by mining operations. However, under SMCRA, states were given primacy to set their own regulatory standards for surface mining as long as they are "no less effective than" Federal standards set by the CWA (Craynon et al. 2012). Reclamation standards of post-mine lands are also enforced under SMCRA (Starnes and Gasper 1995). Since the enactment of these legislative measures, much scientific focus has been placed on monitoring efforts to regulate mining activities, environmental impacts, and reclamation of post-mined lands (Starnes and Gasper 1995). However, data is not currently sufficient to predict the spatial impact of mining on stream conditions, nor is there a good understanding of the scope of subsequent changes in biological structure and function in aquatic ecosystems in watersheds where mining occurs (Simmons et al. 2008). Fully understanding the impacts of coal mining to central Appalachian streams is critical to develop proper reclamation/mitigation techniques.

COAL MINING IMPACTS

Currently, the extraction of Appalachian coal by mountaintop mining is common as it is cost-effective and allows for mining of shallow coal seams that cannot safely be mined by underground methods (Craynon et al. 2012). Although mountaintop mining accounts for 37% of coal production in Appalachia (National Mining Association 2012), it is the most controversial because of its disproportionately large environmental footprint. Mountaintop coal extraction uses explosives to blast rock layers (overburden) that sit above relatively shallow coal seams (Griffith et al. 2012). This process causes overburden to swell and take up much more volume as haul-able rock pieces than the original compacted geologic material. The amount of overburden produced is too costly to haul off the mine site for disposal, so overburden typically is pushed into nearby valleys. This process exposes vast quantities of unweathered geological materials that when exposed to atmospheric conditions and surface run-off often become acid-forming and/or leach heavy metals along with high levels of salts (Griffith et al. 2012).

Although it is clear that small headwater streams are buried during the mining process altering physical stream habitat, water chemistry, and aquatic biota, the longitudinal spatial scale of indirect disturbance is more challenging to empirically describe. The United States Environmental Protection Agency (EPA) estimates that in central Appalachia, ten percent of the region's land area (500,000 ha) has been affected by mountaintop mining, and over 2,000 km of headwater stream habitat buried by valley fills (EPA 2011). Coal mining is considered a point source pollutant by regulatory agencies such as the EPA and the Virginia Department of Environmental Quality (VADEQ). However, the larger sizes of many surface mines along with the hydrologic changes that occur often require nonpoint source management strategies (Simmons et al. 2008).

Along with direct physical stream loss from valley fills, typically watershed hydrology, stream chemistry, and biological communities also are altered from pre-mined conditions (Starnes and Gasper 1995; Pond et al. 2008; Griffith et al. 2012). Physical alteration of streams resulting from the mining process such as dredging, burial of headwaters, and dam construction clearly alter stream hydrology. Changes from surface coal mining on upland terrestrial processes such as clearing of vegetation, increased impervious surfaces, and changes in topography, can also alter stream hydrology (Simmons et al. 2008; Merriam et al. 2011).

IMPACTS OF OTHER LAND USES

Many of these hydrological changes are not unique to coal mining land use, and are often seen in streams with urban development, and to a lesser extent agricultural and forestry, landuses as well. For example, research in both urban and mining watersheds suggests that as watersheds become increasingly altered, vegetation is replaced by impervious surfaces such as parking lots or coal processing areas, the flow regime (hydrology) of a stream system responds more quickly to rain events becoming "flashy," and results in more frequent and intense stream flood events (Starnes and Gasper 1995; Price et al. 2006). These high flow events can lead to channelization of the stream along with eroded, unstable stream banks (Booth et al. 2004). Instream habitat is buried by sediment as terrestrial erosion continues. If erosion is not controlled, terrestrial processes can be significantly altered by the loss of soil and nutrients (Baker 1985). These dramatic terrestrial changes alter physical stream habitat, and directly and indirectly influence water chemistry. All of these factors have had both acute and chronic effects on specific aquatic biota in mining-influenced streams (Northingham et al. 2011; Pond 2012).

Stream water changes are in part a function of upland landuse alterations caused by practices such as mining, logging, and urbanization within the watershed. In central Appalachia these terrestrial alterations degrade water quality conditions ranging from elevated sediment rates, total dissolved solid (TDS) levels, acid mine drainage (AMD), and heavy metal concentrations in mined watersheds to increases in nutrients and organic matter from improperly discharged wastewater from residential areas (Simmons et al. 2008). The largest (by volume) and most pervasive stream pollutant in the U.S. is stream sediment, which can impact stream biota in two ways: streambed sedimentation and suspended sediment in the water column (Sweeten and McCreedy 2002).

Streambed sedimentation is one of the most prevalent headwater stream pollutants in central Appalachia (Angradi 1999), and occurs when heavier soil particles settle thereby clogging, covering, and embedding in-stream habitat (Harding et al. 1998). Many aquatic species rely on interstitial spaces between rocks and rocky surfaces for breeding, feeding, and predator avoidance. These rock surfaces also provide important attachment substrate for algae, benthic macroinvertebrates, and freshwater mussels. As interstitial spaces fill and streambed substrate covered, many aquatic taxa including benthic macroinvertebrates, freshwater mussels, algae, salamanders, and fishes are negatively affected. Suspended sediment from smaller silt- and clay-sized particles can remain suspended in the water column for extended periods, reducing visibility and light penetration (Matter and Ney 1981; Hartman et al. 2005). Primary producers such as algae and sight-feeding larval fishes such as brook trout (*Salvelinus fontinalis*) found in the headwaters of central Appalachia have been shown to be particularly sensitive to suspended sediments (Marschall and Crowder 1996; Sweeten and McCreedy 2002). In some areas of the central Appalachians, such as northern West Virginia and south-central Pennsylvania, streams have been heavily impacted by AMD from mine runoff and/or discharge that cause stream pH far below natural conditions. AMD is estimated to effect over 4,000 km of streams in Appalachia, and currently it is the single largest non-point source pollutant in Pennsylvania (Reinhardt 1999).

Much is still unknown about many of the terrestrial and aquatic ecosystem dynamics and interactions caused by dramatic landscape alterations from coal mining practices (Stout and Wallace 2005; Simmons et al. 2008). Measuring landuse impacts on stream health is further complicated by the additive effects caused by multiple landuse practices such as agriculture, forest management, and residential areas (Welsh et al. 2005; Morgan et al. 2012). Studies of aquatic macroinvertebrate communities in central Appalachia found that mining stream impacts along with improper wastewater discharge from residential areas had an additive effect on species richness and diversity (Merovich and Petty 2007; Passmore and Pond 2009; Merriam et al. 2011).

CLEAN WATER ACT REQUIREMENTS

In order to regulate and mitigate many of these negative impacts to streams in coal mined watersheds, Section 303(d) of the Clean Water Act requires each state to identify water bodies with degraded water quality, and to set geographically-based total maximum daily load (TMDL) standards. These TMDL standards are designed to best allocate loadings from multiple pollutant sources so as to not exceed the load capacity of the aquatic system (Barbour et al. 1999). The purpose of this process is to assess stream quality, establish pollution reduction controls, and monitor remediation outcomes. Under SMCRA, each state may establish their own regulatory and TMDL standards as long as they are "no less effective than" Federal standards under the CWA (Craynon et al. 2012). If a state determines a waterbody to be degraded, chemically or biologically or both, the stream must be reported to the EPA and placed on the impaired waters list (303(d)).

In Virginia, the VADEQ is responsible for ensuring the compliance of mining operations under the CWA. In order to determine a stream's impairment level, the VADEQ regularly monitors water chemistry parameters, and developed the Virginia Stream Condition Index (VA-SCI), a multi-metric benthic macroinvertebrate assessment protocol to identify biological impairment of streams (Burton and Gerristsen 2003). This protocol uses eight different metrics that evaluate different levels of pollutant sensitivity of macroinvertebrates, such as total taxa numbers, percent pollutant intolerant taxa, and percent pollutant tolerant taxa, to produce an overall stream condition score ranging from 0 (devoid of life) to 100 (optimal stream life). Based on comparison to local reference stream conditions, the VA-SCI places any stream scoring below 60 on the EPA impaired waters list (CWA 303(d)).

Once a stream is listed as impaired, the VADEQ uses a three-step TMDL process to improve water quality conditions. Individual stream characteristics and specific impairments such as sedimentation, conductivity, and low pH, are used to set TMDL standards, and an appropriate improvement implementation plan is developed and implemented (MapTech 2008). However, monitoring using the TMDL approach does not always provide sufficient guidance for the development of best management practices to effectively improve water quality and biotic communities. Guidance is particularly lacking in systems heavily impacted by nonpoint source pollution such as surface coal mines (Barbour et al. 1999). Endpoints for removing a stream from the impaired waters list also are established by the VADEQ, varying on a stream-by-stream basis. Often these endpoints are not clearly outlined in TMDL implementation plans (see Map Tech 2004); therefore achieving an unimpaired status can be complicated and problematic. In order to properly manage water resources using TMDL standards, there must be a full understanding of the relationships among water chemistry, effluent sources, non-chemical stressors, chemical stressors, and control practices.

BIOLOGICAL INTEGRITY

Unfortunately, such TMDL standards often lack clear and consistent linkages to bioassessment results and/or biological integrity (Barbour et al. 1999). For example, in laboratory toxicology testing of water with elevated TDS, hardness, and alkalinity levels from two mining-impacted streams in West Virginia were acutely toxic to freshwater daphnia (*Ceriodaphnia dubia*). However, in field studies, these two streams had relatively high richness of macroinvertebrates in the orders considered to be very sensitive to poor water quality conditions: Ephemeroptera, Plecoptera, and Trichoptera (EPT) (Merricks et al. 2007). Without these linkages, TMDL standards may not be accurate. An overestimation of a stream's loading capacity can result in biological degradation, whereas an underestimation may result in time and financial resources wasted on needlessly managing non-impactful chemical parameters. Inherent stream variability also may contribute to the complexity of interactions and linkages that must be understood to develop accurate TMDL standards.

As linear ecosystems, streams exhibit high levels of natural spatial and temporal variability influenced by both stochastic and anthropogenic instream, atmospheric, and watershed changes (Cummins 1975). Appalachian headwater streams are estimated to account for up to 75% of stream mileage in this region (Peterman and Semlitsch 2009). These headwater streams are closely linked to proximal terrestrial processes, have little to no photosynthetic production, and are diverse in benthic macroinvertebrate communities and generally dominated by shredder and collector taxa (Cummins 1975). As stream order increases, the direct influence of adjacent terrestrial habitat lessens and the influence of cumulative discharge from the multiple upstream drainages becomes more significant (Cummins 1975). This river continuum has been well-accepted by the scientific community to explain general changes in biological assemblages and physical habitat along the stream gradient (Vannote et al. 1980). For example, in an evaluation of central Appalachian rivers, Kerans and Karr (1994) recommend that shredder macroinvertebrate abundance should not be used in assessment of higher order streams since natural abundance of this group is not great enough to accurately detect water impairments. However, shredder abundance may be an important metric in headwaters where shredders are more abundant and shown to be an accurate indicator of riparian alterations (Kerans and Karr 1994).

While the stream continuum theory helps to explain and minimize some of the natural variability that occurs in streams, many factors, such as local precipitation events, temporal scales, site-specific conditions, and transient dynamics (or the often unexplained variation in ecosystem behavior from its long-term trends with or without variation in external conditions) should be considered. These natural processes all create difficulty in separating natural, "background" variability from anthropogenic impacts on aquatic systems (Hastings 2004; Hastings 2010; Johnson and Hering 2010). For example, in a survey of rivers throughout central Appalachia, Kerans and Karr (1994) found some metrics such as proportion of chironomids, shredders, and detritivores showed a closer relationship to the temporal scale than to location, indicating that differences in macroinvertebrate assemblages may not always be the result of site quality (Kerans and Karr 1994). In order to best distinguish natural stream variations from anthropogenic impacts, it is important to have long-term stream condition data sets that analyze a variety of biological responses, since stressors may be revealed at various levels of biological organization as well as in physical stream habitat and water quality changes (Adams et. al 2002; Griffith et al. 2012).

Without historical data to provide insight into pre-disturbance conditions (and thereby provide an end goal), less-disturbed, i.e., unmined, reference watersheds are generally used as a surrogate (Younos et al. 2007). Finding a "truly undisturbed watershed" in central Appalachia is difficult as very few, if any, exist, considering the widespread and pervasive nature of documented and undocumented past mining, timbering, and other cultural activities since European settlement (Simmons et al. 2008). Measuring the deviation of abundance and assemblage of taxa between mining impacted and reference streams may be useful in measuring impairment (Merovich and Petty 2007). Comparisons to reference streams also are used often by regulatory agencies (EPA and VADEQ) to identify and monitor stream degradation (Barbour et al. 1999). Ideally reference watersheds should have similar topography, watershed size, geology, and soil types to impaired stream watersheds.

However, scientific standards for the selection of reference watersheds have not been established, and reference site choices are often left to "best professional judgment" (Younos et al. 2007). In central Appalachia, availability of optimal reference sites is limited, forcing water resource managers to select fewer reference sites or choose reference sites with different characteristics from the study watershed (Younos et al. 2007). For example, in the development of the statewide VA-SCI metrics that encompasses six level III ecoregions, a total of 62 reference sites were used; however, only five of these sites were in the central Appalachian ecoregion 69 (Burton and Gerritsen 2003). In a comparative study of mining impacts on headwater streams in southwest Virginia, Matter and Ney (1981) had to choose a 400% larger reference watershed that was located almost 50 km away from the study watersheds. However, even sub-optimal reference watersheds still offer the only insight into the establishment of water quality standards (Younos et al. 2007).

In response to national water quality concerns and monitoring approaches, the EPA Office of Water has advanced the Watershed Protection Approach (WPA) to consolidate and reorient state and Federal agencies that manage and regulate water resources by using watershed level management units. Past management focus has often been placed on small, highly degraded stream sections or individual pollutant sources. Often these efforts only provide a limited view of the total extent of degradation on a watershed with multiple mines or land uses (Barbour et al. 1999; Freund and Petty 2007). Ideally, the watershed approach provides an integrated and comprehensive evaluation of ecological conditions (integrity) in coal mining areas in order to best and most effectively protect aquatic resources (Barbour et al. 1999).

In central Appalachia, mining impacts are often assessed and mitigated on an individual mine basis. Using the watershed approach helps to better manage the longitudinal, additive effects of multiple land uses such as mining and residential wastewater on stream health (Freund and Petty 2007). Common in mine-impacted watersheds, altered stream chemistry from AMD and/or high conductivity can isolate biological communities in upstream reaches reducing genetic diversity and recolonization ability. Linking upstream and downstream biological communities is therefore particularly important for the protection of the many endemic species found in central Appalachia. These linkages can most effectively be created and maintained when managing water resources at the watershed level (Freund and Petty 2007). The watershed approach allows for a cost-effective, realistic management planning where focus can be placed on restoration efforts for larger sections of less impacted streams instead of wasting resources on small, highly degraded streams that may never recover (Freund and Petty 2007).

Although there is an increased effort to remediate stressors within watersheds using the watershed approach, full understanding of how to measure stream conditions in the context of post-mining recovery continues to be a highly debated topic among scientists, regulators, and mining companies (Kerans and Karr 1994; Harding et al. 1998; Allan 2004; Milman and Short 2008; Simmons et al. 2008). The EPA and many states have developed detailed techniques for assessing streams over a large spatial scale, such as a region or state (i.e., EPA Rapid Bioassessment Protocol, VA-SCI and the West Virginia Stream Condition Index or WV-SCI). For example, the VADEQ uses the same protocol and scoring metrics (VA-SCI) on streams impacted by coal mining in the Appalachian Plateau region of southwest Virginia as are used for agricultural or urban-impacted streams in the Piedmont and Coastal Plain.

DISCUSSION

Often to assess recovery and restoration, focus is placed on one stream condition parameter such as water chemistry or benthic macroinvertebrates over a relatively short time period (Kerans and Karr 1994). This approach often provides limited insight into the full extent of recovery and restoration success. Due to the natural temporal and spatial variability of stream ecosystems and the impacts of multiple anthropogenic stressors in central Appalachia on stream ecosystems,

parameter choice can be somewhat problematic (Adam et al. 2002; Carlisle et al. 2009). The questions that remain are relatively simple to frame, but more difficult to answer: What is the level of aquatic ecosystem degradation? What spatial remediation is required to reach a valid benchmark of improvement? What factors should these benchmarks include?

Biological sampling "directly measures the condition of the resource at risk, detects problems that other methods may miss or underestimate, and provides a systematic process for measuring progress resulting from the implementation of water quality programs" (EPA 1990). Although not a substitute for chemical and toxicological monitoring, biological assemblages seem to show responses to the wide range of chemical conditions in mining impacted streams, even revealing additive effects of pollutants that individually do not exceed regulatory standards (Karr 1991; Freund and Petty 2007). Aquatic biological communities are sensitive to a range of environmental factors and watershed conditions across both long- and short-term temporal scales (Karr 1981). In central Appalachian coalfields, agency monitoring programs as well as research efforts often have used single taxon assessment protocols that usually focus on benthic macroinvertebrates such as the VA-SCI and WVSCI (Stout and Wallace 2005; Santos and Stevenson 2011; Hartman et al. 2005; Pond 2008; Passmore and Pond 2009; Merricks et al. 2007). Instead of focusing on a single group of organisms to monitor biological degradation, the integration of multiple taxa into bioassessment protocols has been shown to better measure both aquatic impairment and restoration status (Adams et al. 2002; Freund and Petty 2007).

In east Tennessee, Adams et al. (2002) found that using a suite of taxa such as periphyton, macroinvertebrates, and fish, improved the ability to identify the status of ecological recovery of disturbed aquatic systems. Biotic recovery often occurs on various time and spatial scales. Recolonization rates of biota may be affected by factors such as physiochemical stream conditions, taxa dispersal method, and pollution tolerance levels. Therefore, using a single taxon or trophic level for stream bioassessment may not always reflect a holistic ecosystem recovery. Incorporating bio-indicators from multiple levels of biological organizations and trophic levels helps to ensure a full view of recovery processes (Adams et al. 2002).

Although often considered to be too expensive and time-intensive, biomonitoring protocols have been established by state agencies such as the Ohio EPA and Federal agencies such as the EPA. For example, the Ohio EPA biomonitoring protocols incorporate benthic macroinvertebrates, fish, and salamanders that identify stream permanence as well as biological degradation (Ohio EPA 2012). The EPA's Rapid Bioassessment Protocol (Barbour et al. 1999) outlines empirical, yet cost-effective protocols for biomonitoring multiple stream taxa including periphyton, benthic macroinvertebrates, and fish. In order to select biomonitoring taxa that will accurately and cost-effectively detect aquatic degradation in a given region, it is important to fully understand each taxa's strengths and weaknesses for indicating environmental changes.

In central Appalachia, periphyton (algae) has been used infrequently in conjunction with water quality and/or macroinvertebrate surveys of mining impacted streams (Merricks et al. 2007). Primary producers such as periphyton can be directly affected by alterations in physical and chemical parameters, and often show significant responses when compared to other taxa at lower concentrations of stressors, such as herbicides (Stevenson 1998; Stevenson and Pan 1999; Barbour et al. 1999). Due to their short lifespan and quick reproductive rates, periphyton can be excellent indicators for examination of short-term impacts and recovery of degraded water quality. Algae are easy, quick, and cost-effective to sample and require little field training, and sampling is minimally invasive to the stream. However, laboratory analysis can be time consuming and requires specialized knowledge/training, although well-established, standard community assessment protocols do exist (Barbour et al. 1999). Periphyton biomass and benthic macroinvertebrate assemblages have been shown to have close linkages in response to sedimentation and nutrient loading in large streams (Molinos and Donohue 2010) as well as to AMD impacts (Meegan and Perry 1996).

In central Appalachia, benthic macroinvertebrates have been relied on heavily for bioassessment for a variety of reasons. First, benthic macroinvertebrates have been shown to serve well as a site-specific, short-term biological indicator. Many trophic- and pollution-tolerance levels are represented in this group, and sampling is relatively easy and inexpensive. Invertebrates are also present in headwater streams that have limited fish communities (Barbour et al. 1999). Although bioassessment using benthic macroinvertebrates has many advantages, large variations in seasonal assemblages, sampling location within a watershed, and analysis of data often causes conflicting and unclear results (Kerans and Karr 1994; Fulk et al. 2003). Seasonal variability of macroinvertebrate assemblages has been debated with little resolution. Fulk et al. (2003) and Timpano et al. (2011) found seasonal variability between spring and fall macroinvertebrate scores when using the WV-SCI and VA-SCI assessment protocols, respectively. However, in the VA-SCI development and field validations, no seasonal differences in invertebrate scores were shown (Burton and Gerritsen 2003). If macroinvertebrate assemblages show significant seasonal variations, the VA-SCI protocol scoring should be adjusted in order to account for seasonality differences (Timpano et al. 2011). Currently the VA-SCI score has no such allowances for temporal variabilities and scores below 60 are considered impaired regardless of season (Timpano et al. 2011).

Sampling location selection at both the regional and local levels has also been shown to produce variability in macroinvertebrate community structures. The VADEQ found that reference streams in the coalfields of southwestern Virginia tended to have lower scores using the VA-SCI assessment method as compared to other ecoregions in Virginia. This may be due to natural differences in macroinvertebrate assemblages of central Appalachian streams relative to those of other Virginia ecoregions (Burton and Gerritsen 2003). The Ohio EPA recommends designing quantitative biological criteria based on individual ecoregion characteristics (Yoder 1991). The VA-SCI attempts to parsimoniously fit metrics that will work for all six ecoregions in Virginia. However, this approach may sacrifice some accuracy and resolution on the ecoregion level in order to encompass the whole state with one bioassessment protocol.

Macroinvertebrate assemblages may vary at different stream order and permanence levels within a watershed. In the southern Appalachians, Feminella (1996) found little variation in community structure between intermittent first order streams and permanent lower order streams. Santos and Stevenson's (2011) results from a Massachusetts study were similar to Feminella's, whereas Brown and Brussock (1991) found lower macroinvertebrate diversity in intermittent headwater streams in the Ozarks. Although these studies were conducted in different regions of the United States, they do indicate possible variability in macroinvertebrate assemblages along stream permanence gradients. Moreover, watershed size has been shown to influence macroinvertebrate assemblages and abundances as well. In central Appalachia, macroinvertebrate species richness and abundance, particularly the number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa, increase rapidly along the stream gradient until watershed size of about 100 acres, where the rate of increase becomes constant (Stout and Wallace 2005). The percentage of collector macroinvertebrates also increased along this gradient whereas percent shredders decreased (Stout and Wallace 2005).

Without resolution to the scope/scale of these natural temporal and spatial influences on macroinvertebrate assemblages, it is extremely difficult to properly account for natural variation within biomonitoring protocols such as the VA-SCI and WV-SCI. Without a full understanding of these natural variations, there may be inaccuracies in assessment of macroinvertebrate abundance and diversity. Analysis of macroinvertebrate surveys varies greatly among agencies and biologists, ranging from traditional EPT evaluation to multiple-metric analysis of community function and structure (Blocksom and Winters 2006). Taxonomic identification level of macroinvertebrates has not been standardized among regulatory agencies, researchers, and managers. Hilsenhoff (1987) created a simple index of biotic diversity by assigning pollution tolerance values to macroinvertebrates at the genus-level and dividing the sum of the tolerance values by the total number of individuals identified (around 100). In order to address time and fiscal constraints faced by many management agencies, Hilsenhoff (1988) used 120 Wisconsin streams to develop a family-level version of the index that reduced assessment time by 50%. However, the family-level index lost some resolution compared to genus-level evaluation. Although Pond et al. (2008) did not find significant differences in bioassessment results between identification to the genus-level and identification to the family-level, genus-level indices were recommended for measuring stream conditions in mined watersheds. These identification level differences can make direct comparison of bioassessment results analyzing regional and temporal trends in macroinvertebrate assemblages and coal mining impacts difficult.

Due to their high mobility and relative longevity, fish can serve as good long-term indicators of chronic stressors and reflect broad-habitat conditions of central Appalachian streams (Matter and Ney 1981; Angermeier and Karr 1986; Fulk et al. 2003; Freund and Petty 2007). Fish account for almost half of the endangered vertebrate species in the United States (Warren and Burr 1994) and represent a wide range of trophic levels and life histories (Angermeier and Karr 1986). A study of mine-impacted streams in West Virginia that compared indicator characteristics of fish and benthic macroinvertebrates found fish communities to be less responsive than macroinvertebrates to degraded water quality, suggesting that fish may not be a good indicator for bioassessment of local conditions (Freund and Petty 2007).

However, Birge et al. (2000) observed that fish were good biosensors for heavy metal levels in Kentucky streams, and may more accurately reflect watershed conditions than macroinvertebrates. In addition, the life history and environmental requirements of most fish species are well understood, allowing for better understanding of community structure and function (Fausch et al. 1990). Most stakeholders, policy makers and land managers are more familiar and knowledgeable about fish (especially game species) than macroinvertebrates, thereby making fish assessment results more relevant and relatable (Karr 1981). Compared to other taxa, sampling of fish communities is cost-effective, and requires relatively little training or laboratory time (Karr 1981; Fausch et al. 1990). However, as with many taxa, sampling can be quite labor intensive and identification requires extensive knowledge. Fish communities are often influenced by stream permanence and by both diel and seasonal variations (Karr 1981). Fish assemblages may also be limited due to physical or chemical barriers (such as AMD) lower in the watershed.

Amphibians, particularly salamanders, are an important component to both terrestrial and aquatic ecosystems in central Appalachia (Davic and Welsh 2004). Long-lived stream salamander species have also been recommended for use as an indicator for aquatic bioassessment protocols due to their natural high abundances in undisturbed areas, sensitivity to stream and watershed degradation, and role as a keystone faunal group (Welsh and Ollivier 1998; Welsh and Droege 2001; Southerland et al. 2004). In headwater streams absent of fish, salamanders are the dominant, most abundant vertebrate predator. Stream salamanders provide a significant intermediate role in trophic cycling, feeding on small prey such as benthic macroinvertebrates and, in turn, are an important prey item for larger vertebrates (Petranka et al. 1993; Davic and Welsh 2004; OHEPA 2012). Central Appalachian riparian areas typically have relatively stable salamander populations (compared to macroinvertebrates), with densities of up to $1.4/m^2$ (Kleeberger 1984; Hairston and Wiley 1993; Welsh and Ollivier

1998). Life history and behavior vary greatly among salamander species with some larval forms having long aquatic phases of up to 48 months, whereas other species have very short- or even no aquatic larval phase (OHEPA 2012). Due to such variation in life history, the Ohio EPA uses larval salamander surveys as a method for accurately and cost-effectively indicating stream permanence and headwater habitat quality. Evidence of reproduction (eggs and larvae) of salamander species with larval phases exceeding 12 months has been found to indicate perennial stream flow, while presence of species with larval phases of less than 12 months indicates intermittent stream flow (Ohio EPA 2012).

Salamander species that move between aquatic and terrestrial environments provide an important link between upland and stream ecosystems (Fisher et al. 1998). Stream salamanders are considered to be sensitive to aquatic stressors and environmental degradation, and low stream salamander abundance has been shown to be closely linked to terrestrial watershed degradations (Petranka et al. 1993; Willson and Dorcas 2003). Stream salamander abundances have been shown to be negatively impacted, even to the point of local extirpation, by decreases in stream conditions such as streambed sedimentation, temperature, pH, and suspended sediment, as from watershed land uses such as mining, urbanization, and timber harvesting (Gore 1983; Willson and Dorcas 2003; Welsh et al. 2005; Moseley et al. 2008). This gradient of response to anthropogenic impacts is critical for an indicator taxa group. If a group of organisms is highly sensitive to environmental degradation they will be quickly extirpated from the area, and may only indicate mild water quality impairments (Welsh and Ollivier 1998).

Taxa that are extremely tolerant of impaired stream conditions should also not be selected for use in biomonitoring as they will not show a response until the aquatic ecosystem is heavily degraded. Movement of many Appalachian stream salamander species has been shown to be less than 100m (Pauley et al. 2000), and although stream salamanders are able to cross barriers such as dry streambed sections and waterfalls, mobility may be limited by riparian and upper watershed habitat fragmentation (Resetarits 1997; Williams et al. 2002; Willson and Dorcas 2003; OHEPA 2012). Stream salamanders may also be useful to indicate recovery endpoints as their low mobility reduces the likelihood of increasing abundances being a reflection of immigration, and helps to confirm that local, long-term stream and watershed conditions are suitable for a positive rate of salamander survival and reproduction (Welsh and Ollivier 1998; Welsh et al. 2005).

CONCLUSION

Traditionally, research and monitoring of aquatic ecosystems has focused on the effects of land use practice disturbances. However, the ability to distinguish natural stream variability from the additive effects of anthropogenic influences is currently limited. Furthermore, there is little known about the multiple temporal and spatial dynamics of the biological recovery of a degraded aquatic system (Harding et al. 1998; Adams et al. 2002; Niemi et al. 2004; Hartman et al. 2005; Milman and Short 2008; Simmons et al. 2008; Northingham et al. 2011). Currently, even identifying and quantifying aquatic ecosystem structure and function of undisturbed systems can be very difficult (Simmons et al. 2008). Fiscal and personnel constraints are often limiting factors to widespread or more comprehensive sampling. It is, however, imperative to establish scientifically-backed bench marks of success for watershed remediation efforts and/or coal mine reclamation. Biological assessment using a suite of taxa may provide the most extensive view of aquatic changes including degradation and recovery. In order to develop optimal bioassessment protocols, it is essential to first gain a better understanding the response of each taxa to water conditions as well as the relationship dynamics among taxa. Investigation of salamanders as bio-indicators for streams in mined watersheds should be a priority, since little is known about their indicator abilities in central Appalachia.

ACKOWLEDGMENT

This study was sponsored by the Appalachian Research Initiative for Environmental Science (ARIES). The views, opinions and recommendations expressed herein are solely those of the authors and do not imply any endorsement by ARIES employees, other ARIES-affiliated researchers or industrial members. Information about ARIES can be found at http://www.energy.vt.edu/ARIES.

REFERENCES

Adams, S.M., W.R. Hill, M.J. Peterson, M.G. Ryon, J.G. Smith and A.J. Stewart. 2002. Assessing recovery in a stream ecosystem: Applying multiple chemical and biological endpoints. *Ecological Applications*. 12:1510–1527.

Ahlstedt, S.A., M.T. Fagg, R.S. Butler, and J.F. Connell. 2005. *Long-term trend information for freshwater mussel populations at twelve fixed-station monitoring sites in the Clinch and Powell rivers of eastern Tennessee and southwestern Virginia, 1979–2004.* Unpublished Report, U.S. Fish and Wildlife Service, Cookeville, Tennessee.

Allan, J.D. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annual Review of Ecology Evolution and Systematics*. 15:257–284.

Angermeier, P.L. and J.R. Karr. 1986. Applying an index of biotic integrity based on stream-fish communities: Considerations in sampling and interpretation. *North American Journal of Fisheries Management*. 6:418–429.

Angradi, T.R. 1999. Fine sediment and macroinvertebrate assemblages in Appalachian streams: A field experiment with biomonitoring applications. *Journal of the North American Benthological Society*. 18:49–66.

Baker, D.B. 1985. Regional water quality impacts of intensive row-crop agriculture: A Lake Erie Basin case study. *Journal of Soil and Water Conservation*. 40:125–132.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. *Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, Second Ed.* EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C. 339 pages.

Bernhardt, E.S. and M.A. Palmer. 2011. The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the Central Appalachians. Evaluation Studies Review. *Annual New York Academy of Science*. 1223:39–57.

Birge, W.J., D.J. Price, J.R. Shaw, J.A. Spromberg, A.J. Wigginton and C. Hogstrandt. 2000. Metal body burden and biological sensors as ecological indicators. *Environmental Toxicology and Chemistry*. 4:1199–1212.

Blocksom, K.A. and L. Winters. 2006. *The evaluation of methods for creating defensible, repeatable, objective and accurate tolerance values for aquatic taxa.* U.S. Environmental Protection Agency, Office of Research and Development, Cincinnati, OH. 52 pages.

Booth, D.B., J.R. Karr, S. Schauman, C.P. Konrad, S.A. Morley, M.G. Larson and S.J. Burges. 2004. *Journal of the American Water Resources Association*. 40:1351–1364.

Brussock, P.P. and A.V. Brown. 1991. Riffle-pool geomorphology disrupts longitudinal patterns of stream benthos. *Hydrobiologia*. 220:109–117.

Burton, J. and J. Gerritsen. 2003. *A stream condition index for Virginia non-coastal streams*. Tetra Tech, Inc. Virginia Department of Environmental Quality, Richmond, Virginia. 168 pages.

Carlisle, D.M., J. Falcone and M.R. Meador. 2009. Predicting the biological condition of streams: use of geospatial indicators of natural and anthropogenic characteristics of watersheds. *Environmental Monitoring Assessment*. 151:143–160.

Craynon, J.R., E.A. Sarver and D.P. Robertson. 2012. Could a public ecology approach help resolve the mountaintop mining controversy? *Resources Policy*. http://dx.doi.org/10.1016/j.resourpol.2012.08.001.

Cummins, K.W. 1975. *The ecology of running waters theory and practice*. Kellogg Biological Station. Hickory Corners, MI. 16 pages.

Davic, R.D. and H.H. Welsh. 2004. On the ecological roles of salamanders. *Annual Review of Ecology, Evolution, and Systematics*. 35:405–434.

Fausch, K.D., J. Lyons, J.R. Karr, and P.L. Angermeier. 1990. Fish Communities as indicators of environmental degradation. *American Fisheries Society Symposium*. 8:123–144.

Feminella, J.W. 1996. Comparison of benthic macroinvertebrates assemblages in small streams along a gradient of flow permanence. *Journal of North American Benthological Society*. 15:651–669.

Fisher, S.G., N.B. Grimm, E. Marti, R.M. Holmes and J.B. Jones, Jr. 1998. Material spiraling in stream corridors: a telescoping ecosystem model. *Ecosystems*. 1:19–34.

Freund, J.B. and J.T. Petty. 2007. Response of fish and macroinvertebrate bioassessment indices to water chemistry in a mined Appalachian watershed. *Environmental Management*. 39:707–720.

Fulk, F., B. Autrey, J. Hutchens, J. Gerritsen, J. Burton, C. Cresswell and B. Jessup. 2003. *Ecological assessment of streams in the coal mining region of West Virginia using data collected by the U.S. EPA and environmental consulting firms*. U.S. Environmental Protection Agency. National Exposure Research Laboratory. Cincinnati, OH. 152 pages.

Gore, J.A. 1983. The distribution of Desmognathine larvae (Amphibia: Plethodontidae) in coal surface mine impacted streams of the Cumberland Plateau, USA. *Journal of Freshwater Ecology*. 2:13–23.

Griffith, M.B., S.B. Norton, L.C. Alexander, A.I. Pollard and S.D. LeDuc. 2012. The effects of mountaintop mines and valley fills on the physicochemical quality of stream ecosystems in the central Appalachians: a review. *Science of the Total Environment*. 417–418:1–12.

Hairston, N.G. and R.H. Wiley. 1993. No decline in salamander populations: A twenty-year study in the Southern Appalachians. *Brimleyana*. 18:59–64.

Harding, J.S., E.F. Benfield, P.V. Bolstad, G.S. Helfman and E.B.D. Jones. 1998. Stream Biodiversity: the ghost of land use past. *Proceedings of the National Academy of Science, USA*. 95:14843–14947.

Hartman, K.J., M.D. Kaller, J.W. Howell and J.A. Sweka. 2005. How much does valley fill influence headwater streams? *Hydrobiologia*. 552:91–102.

Hastings, A. 2004. Transients: the key to long-term ecological understanding? *Trends in Ecology and Evolution*. 19:39–45.

Hastings, A. 2010. Timescales, dynamics, and ecological understanding. *Ecology*. 91:3471–3480.

Hibbard, W.R. 1990. *Virginia coal: An abridged history*. Virginia Center for Coal & Energy Research. Blacksburg, VA. 160 pages.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. *The Great Lakes Entomologist*. 20:31–39.

Hilsenhoff, W.L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. *Journal of North American Benthological Society*. 7:65–68.

Hocutt, C.H., R.F. Denoncourt and J.R. Stauffer, Jr. 1978. Fishes of the Greenbrier River, West Virginia, with drainage history of the Central Appalachians. *Journal of Biogeography*. 5:59–80.

Johnson, R.K. and D. Herring. 2010. Spatial congruency of benthic diatom, invertebrate, macrophyte, and fish assemblages in European streams. *Ecological Applications*. 20:978–992.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries*. 6:21–27.

Karr, J.R. 1991. Biological integrity: A long-neglected aspect of water resource management. *Ecological Applications*. 1:66–84.

Kerans, B.L. and J.R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. *Ecological Applications*. 4:768–785.

Kleeberger, S.R. 1984. A test of competition in two sympatric populations of Desmognathine salamanders. *Ecology*. 65:1846–1856.

MapTech, Inc. 2004. *General standard TMDL development for Dump's Creek Russell County, Virginia*. Prepared for the Virginia Department of Environmental Quality and Virginia Department of Mines, Minerals, and Energy, Blacksburg, VA. 184 pages.

MapTech, Inc. 2008. *Dumps Creek implementation plan summary*. Prepared for Virginia Department of Environmental Quality, Blacksburg, VA. 28 pages.

Marschall, E.A. and L.B. Crowde. 1996. Assessing population responses to multiple anthropogenic effects: A case study with brook trout. *Ecological Applications*. 6:152–167.

Matter, W.J. and J.J. Ney. 1981. The impact of surface mine reclamation on headwater streams in southwest Virginia. *Hydrobiologia*. 78:63–71.

Meegan, S.K. and S.A. Perry. 1996. Periphyton communities in headwater streams of different water chemistry in the Central Appalachian Mountains. *Journal of Freshwater Ecology*. 11:247–256.

Merovich, G.T. and J.T. Petty. 2007. Interactive effects of multiple stressors and restoration priorities in a mined Appalachian watershed. *Hydrobiologica*. 575:13–31.

Merriam, E.R., J.T. Petty, G.T. Merovich, J.B. Fulton and M.P. Strager. 2011. Additive effects of mining and residential development on stream conditions in a central Appalachian watershed. *Journal of the North American Benthological Society*. 30:399–418.

Merricks, T.C., D.S. Cherry, C.E. Zipper, R.J. Currie and T.W. Valenti. 2007. Coal-mine hollow fill and settling pond influences on headwater streams in southern West Virginia, USA. *Environmental Monitoring Assessment*. 129:359–378.

Milman, A. and A. Short. 2008. Incorporating resilience into sustainability indicators: An example for the urban water sector. *Global Environmental Change*. 18:758–767.

Molinos, J.G. and I. Donohue. 2010. Interactions among temporal patterns determine the effect of multiple stressors. *Ecological Applications*. 20:1794–1800.

Morgan, R.P., K.M. Kline, M.J. Kline, S.F. Cushman, M.T. Sell, R.E. Weitzell and J.B. Churchill. 2012. Stream conductivity: Relationships to landuse, chloride, and fishes in Maryland streams. *North American Journal of Fisheries Management*. 32:941–952.

Morse, J.C., B.P. Stark and W.P. McCafferty. 1993. Southern Appalachian streams at risk: implications for mayflies, stoneflies, caddisflies, and other aquatic biota. *Aquatic Conservation*. 3:293–303.

Moseley, K.R., W.M. Ford, J.W. Edwards and T.M. Schuler. 2008. Long-term partial cutting impacts on *Desmognathus* salamander abundance in West Virginia headwater streams. *Forest Ecology and Management*. 254:300–307.

National Mining Association. 2012. U.S. coal production by state, region and method of mining—2011.

National Mining Association Factsheet. http:// http://www.nma.org/pdf/c_production_mines_state_type.pdf. Accessed January 15, 2013.

Niemi, G.J. and M.E. McDonald. 2004. Application of ecological indicators. *Annual Review of Ecology, Evolution, and Systematics*. 35:89–111.

Northingham, R.M., E.F. Benfeld, S.H. Schoenholtz, A.J. Timpano, J.R. Webster and C. Zipper. 2011. An assessment of structural attributes and ecosystem function in restored Virginia coalfield streams. *Hydrobiologia*. 671:51–63.

Nowacki, G.J. and M.D. Abrams. 1997. Radial-growth averaging criteria for reconstruction disturbance histories from presettlement-origin oaks. *Ecological Monographs*. 67:225–249.

Ohio EPA. 2012. *Field evaluation manual for Ohio's primary headwater habitat streams*. Version 3.0. Ohio EPA Division of Surface Water, Columbus, Ohio. 117 pp.

Ortmann, A.E. 1918. The nayades (freshwater mussels) of the upper Tennessee drainage. *Proceedings of the American Philosophical Society* 57:521–626.

Passmore, M. and G. Pond. 2009. *Evaluating appropriate existing and designated uses of Straight Creek (Lee County, VA) using current macroinvertebrate, habitat, and water quality data*. United States Environmental Protection Agency, Region III, Wheeling, WV. 39 pages.

Pauley, T.K., J.C. Mitchell, R.R. Buech and J. Moriarty. 2000. Ecology and management of riparian habitats for amphibiands and reptiles. In *Riparian Management in Forests of the Continental Eastern United States*, ed. E.S. Verry, J.W. Hornbeck, C.A. Dolloff. pp. 169–91. Boca Raton, FL. Lewis. 402 pages.

Peterman, W.E. and R.D. Semlitsch. 2009. Efficacy of riparian buffers in mitigating local population declines and the effects of even-aged timber harvest on larval salamanders. *Forest Ecology and Management*. 257:8–14.

Petranka, J.W., M.E. Eldridge and K.E. Haley. 1993. Effects of timbering harvesting on southern Appalachian salamanders. *Conservation Biology*. 7:363–370.

Pond, G.J., M.E. Passmore, F.A. Borsuk, L. Reynolds and C.J. Rose. 2008. Downstream effects of mountaintop coal mining: comparing biological conditions using family- and genus-level macroinvertebrate bioassessment tools. *Journal of the North American Benthological Society*. 27:717–737.

Pond, G.J. 2012. Biodiversity loss in Appalachian headwater streams (Kentucky, USA): Plecoptera and Trichoptera communities. *Hydrobiologia*. 679:97–117.

Price, S.J., M.E. Dorcas, A.L. Gallant, R.W. Klaver and J.D. Willson. 2006. Three decades of urbanization: Estimating the impact of land-cover change on stream salamander populations. *Biological Conservation*. 133:436–441.

Reinhardt, C.H. 1999. Acid mine drainage in Pennsylvania streams: "Ironing out" the problem. *Restoration and Reclamation Review*. 5:1–10.

Resetarits, W.J., Jr. 1997. Differences in an ensemble of streamside salamanders (Plethodontidae) above and below a barrier to brook trout. *Amphibia-Reptilia*. 18:15–25.

Santos, A.N. and R.D. Stevenson. 2011. Comparison of macroinvertebrate diversity and community structure among perennial and non-perennial headwater streams. *Northeastern Naturalist.* 18:7–26.

Simmons, J.A., W.S. Currie, K.N. Eshleman, K. Kuers, S. Monteleone, T.L. Negley, B.R. Pohlad and C.L. Thomas. 2008. Forest to reclaimed mine land use change leads to altered ecosystem structure and function. *Ecological Applications.* 18:104–118.

Southerland M.T., R.E. Jung, D.P. Baxter, I.C. Chellman, G. Mercurio and J.H. Vølstad. 2004. Stream salamanders as indicators of stream quality in Maryland, USA. *Applied Herpetology.* 2:23–46.

Starnes, L.B. and D.C. Gasper. 1995. Effects of surface mining on aquatic resources in North America (Revised). *Fisheries.* 20:20–23.

Stein, B.A., L.S. Kutner, and J.S. Adams. 2000. *Precious heritage: the status of biodiversity in the United States.* Oxford, U.K.: Oxford University Press. 399 pages.

Stevenson, R.J. 1998. Diatom indicators of stream and wetland stressors in a risk management framework. *Environmental Monitoring and Assessment.* 51:107–118.

Stevenson, R.J. and Y. Pan. 1999. Assessing ecological conditions in rivers and streams with diatoms. Pages 11–40 in E.F. Stoermer and J.P. Smol, editors. *The diatoms: Applications to the Environmental and Earth Sciences.* Cambridge University Press, Cambridge, UK.

Stout, B. and J.B. Wallace. 2005. *A survey of eight major aquatic insect orders associated with small headwater streams subject to valley fills from mountaintop mining. Mountaintop mining/valley fills in Appalachia: Final Programmatic Environmental Impact Statement.* U.S. Environmental Protection Agency, Washington, D.C. 13 pages.

Sweeten, J. and C.D. McCreedy. 2002. *Suspended sediment: An environmental indicator for warmwater streams.* Technical report for Indiana Department of Environmental Management, Office of Water Management, Indianapolis, IN ARN 98–175.

Timpano, A., S. Schoenholtz, C. Zipper and D. Soucek. 2011. *Levels of dissolved solids associated with aquatic life effects on headwater streams of Virginia's Central Appalachia Coalfield Region.* Virginia Tech, Water Resources Research Center, Blacksburg, VA. 82 pages.

U.S. EPA (Environmental Protection Agency). 1990. *Biological criteria: National program guidance for surface waters.* U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Washington, D.C. EPA 440-5-90-004. 61 pages.

U.S. EPA (Environmental Protection Agency). 2011. *The effects of mountaintop mines and valley fills on aquatic ecosystems of the Central Appalachian coalfields.* Office of Research and Development, National Center for Environmental Assessment, Washington, D.C. 136 pages.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell and C.E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences.* 48:2477–2480.

Warren, M.L., Jr., and B.M. Burr. 1994. Status of freshwater fishes of the US: Overview of an imperiled fauna. *Fisheries.* 19:6–18.

Welsh, H.H. and S. Droege. 2001. A case for using plethodontid salamanders for monitoring biodiversity and ecosystem integrity of North American forests. *Conservation Biology* 15:558–569.

Welsh, H.H. and L.M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: A case study from California's redwoods. *Ecological Applications.* 8:1118–1132.

Welsh, H.H., G.R. Hodgson and A.J. Lind. 2005. Ecogeography of herpetofauna of a northern California watershed: Linking species patterns to landscape processes. *Ecography.* 28:521–536.

Wilcove, D.S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States. *BioScience.* 48:607–615.

Williams, L.R., D.L. Crosswhite and M.G. Williams. 2002. Short-term effect of riparian distribution on *Desmognathus brimleyorum* at a natural spring in Oklahoma. *Southwest Naturalist.* 47:611–613.

Willson, J.D. and M.E. Dorcas. 2003. Effects of habitat disturbance on stream salamanders: Implications for buffer zones and watershed management. *Conservation Biology.* 17:763–771.

Yoder, C.O. 1991. *Answering some concerns about biological criteria based on experiences in Ohio.* Biological Criteria: Research and Regulation, Proceedings of a National Conference, U.S. EPA, Office of Water, Washington, D.C. 8 pages.

Younos, T., G. Yagow, L. Shabman, L. Smock, E. Smith, H. Kator and J. Walker. 2007. *Protocols for reference watershed approach.* Virginia Department of Environmental Quality, Academic Advisory Committee. 7 pages.