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The Effects of Mountaintop Mines and Valley Fills on Aquatic Ecosystems of the Central Appalachian Coalfields

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ABSTRACT

This report assesses the state of the science on the environmental impacts of mountaintop mines and valley fills (MTM-VF) on streams in the Central Appalachian Coalfields. These coalfields cover about 48,000 square kilometers (12 million acres) in West Virginia, Kentucky, Virginia and Tennessee, USA. Our review focused on the impacts of mountaintop removal coal mining, which, as its name suggests, involves removing all or some portion of the top of a mountain or ridge to expose and mine one or more coal seams. The excess overburden is disposed of in constructed fills in small valleys or hollows adjacent to the mining site.

Our conclusions, based on evidence from the peer-reviewed literature and from the U.S. Environmental Protection Agency's Programmatic Environmental Impact Statement released in 2005, are that MTM-VF lead directly to five principal alterations of stream ecosystems: (1) springs, intermittent streams, and small perennial streams are permanently lost with the removal of the mountain and from burial under fill, (2) concentrations of major chemical ions are persistently elevated downstream, (3) degraded water quality reaches levels that are acutely lethal to standard laboratory test organisms, (4) selenium concentrations are elevated, reaching concentrations that have caused toxic effects in fish and birds and (5) macroinvertebrate and fish communities are consistently and significantly degraded.

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LIST OF ABBREVIATIONS AND ACRONYMS

AMD	acid mine drainage
AOC	approximate original contour
ARRI	Appalachian Regional Reforestation Initiative
AWQC	ambient water quality criterion
BCG	biological condition gradient
$CaCO_3$	calcite
$CaMg(CO_3)$	dolomite
$CaSO_4$	calcium sulfate
CWA	Clean Water Act
DO	dissolved oxygen
EC10	effect concentration for 10% of the tested organisms
EMAP	Environmental Monitoring and Assessment Program
EPA	U.S. Environmental Protection Agency
EPT	Ephemeroptera, Plecoptera and Trichoptera
FEC20	field-based effect concentration for 20% of the tested organisms
GIS	geographic information system
GLIMPSS	genus-level index of most probable stream status
HBI	Hilsenhoff Biotic Index
IBI	Index of Biotic Integrity
KSO_4	potassium persulphate
LC50	lethal concentration for 50% of the tested organisms
LOEC	lowest-observed-effect concentration
MBI	macroinvertebrate bioassessment index
$MgSO_4$	magnesium sulfate
MHRW	moderately hard reconstituted water
MTM-VF	mountaintop mines and valley fills
NPDES	National Pollutant Discharge Elimination System
OSM	Office of Surface Mining
PAH	polycyclic aromatic hydrocarbons
PEIS	programmatic environmental impact statement
SMCRA	Surface Mining Control and Reclamation Act
SOC	soil organic carbon
TDS	total dissolved solids
USACE	United States Army Corps of Engineers
WV SCI	West Virginia Stream Condition Index

FOREWORD

Headwater streams and watersheds in Appalachia play a disproportionately large role in the region's ecology. They are sources of clean, abundant water for larger streams and rivers, are active sites of the biogeochemical processes that support both aquatic and terrestrial ecosystems, and are characterized by exceptional levels of plant and animal endemism (i.e., biodiversity hotspots). The benefits of healthy headwaters are cumulative as the critical ecological functions of many small streams flowing into the same river system are necessary for maintaining ecological integrity.

The practice of mountaintop mining and valley fills, which has become increasingly common in Appalachian states, can have major environmental consequences for the mountain ecosystem, the nearby valleys and downstream water quality. There is a growing body of evidence in the scientific literature that valley fills from mountaintop mining are having deleterious ecological effects. Recent published reports show that as water quality deteriorates downstream of a valley fill, the biota within the stream are likewise affected.

The mining of coal in the United States is highly regulated. Mountaintop mining, in particular, involves multiple statutes and agencies at both the federal and state levels. The two key federal laws are the Surface Mining Control and Reclamation Act (SMCRA, 25 U.S.C. § 1201) and the Clean Water Act (CWA, 33 U.S.C. § 1252). The key entities at the federal level are the Office of Surface Mining (OSM), the Environmental Protection Agency (EPA) and the U.S. Army Corps of Engineers (USACE). On June 11, 2009, in a Memorandum of Understanding, these agencies committed to a series of activities to improve the regulation of mining practices under existing statutory authorities.

This assessment report is one of several actions EPA has initiated to better understand the ecological impacts of mountaintop mining. For this report, the EPA Office of Research and Development has reviewed and assessed the published peer-reviewed literature on the aquatic impacts associated with mountaintop mining. This version of the assessment will undergo an external peer review by EPA's Science Advisory Board. The final peer-reviewed assessment will inform the EPA as it continues to implement its regulatory duties under the Clean Water Act.

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1. EXECUTIVE SUMMARY

Mountaintop mines and valley fills (MTM-VF) lead directly to five principal alterations to stream ecosystems: (1) springs, intermittent streams, and small perennial streams are permanently lost with the removal of the mountain and from burial under fill, (2) concentrations of major chemical ions are persistently elevated downstream, (3) degraded water quality reaches levels that are acutely lethal to standard laboratory test organisms, (4) selenium (Se) concentrations are elevated, reaching concentrations that have caused toxic effects in fish and birds, and (5) macroinvertebrate and fish communities are consistently and significantly degraded. These conclusions are based on evidence, described in this report, from the peer-reviewed literature and from the U.S. Environmental Protection Agency (EPA) Programmatic Environmental Impact Statement (PEIS) released in 2005. Our review focused on the impacts on mountaintop removal coal mining, which as its name suggests, involves removal of all or some portion of the top of a mountain or ridge to expose and mine one or more coal seams. The excess overburden is disposed of in constructed fills in small valleys or hollows adjacent to the mining site.

Evidence shows that concentrations of chemical ions are, on average, about 10 times higher downstream of MTM-VF than in streams in unmined watersheds. Sulfate (SO_4^{2-}), bicarbonate (HCO_3^-), calcium (Ca^{2+}) and Magnesium (Mg^{2+}) are the dominant ions in the mixture, but potassium (K^+), sodium (Na^+), and chloride (Cl^-) are also elevated. These ions all contribute to the elevated levels of total dissolved solids (TDS), typically measured as specific conductivity, observed in the effluent waters below valley fills. These ions do not degrade or precipitate out of the water column. Concentrations decrease only when diluted by another, cleaner, source of water.

Water from sites having high chemical ion concentrations downstream of MTM-VF is acutely lethal to invertebrates in standard aquatic laboratory tests, and models of ion toxicity based on laboratory results predict that acute toxicity would be expected from the ions alone. Benthic macroinvertebrate assessments of condition frequently score “poor quality” at sites downstream of MTM-VF that have high ion concentrations.

Selenium concentrations are also elevated downstream of MTM-VF. Selenium can bioaccumulate through aquatic food webs, and elevated levels have been found in fish in this mining region. More than half of the sites surveyed downstream of MTM-VF exceeded the chronic Ambient Water Quality Criterion (AWQC) for selenium. Selenium has been associated with increased death and deformities in fish and reduced hatching in birds in studies of coal overburden effluents in other regions.

Permits already approved from 1992 through 2002 are projected, when fully implemented, to result in the loss of 1,944 km of headwater streams. This represents a loss of almost two percent of the stream miles in the focal area (KY, TN, WV, and VA), a length that is more than triple the length of the Potomac River, just during this 10-year-period. We found no studies that updated the MTM-VF inventory conducted as part of the PEIS in 2002, but both mine footprint and stream losses were projected to double by 2012. An updated inventory would allow statistically sound estimates of cumulative stream loss and is a critical information need.

Reclamation practices (e.g., contouring and revegetation) were common in all of the reviewed studies. The data indicate that reclamation partially controls the amount of soil erosion and fine sediments transported and deposited downstream. The acidic drainage that is often associated with coal mining is largely neutralized through reactions with carbonate minerals within the valley fills or treatment in the sediment retention ponds. Yet, because ions, metals, and selenium below MTM-VF were elevated in the reviewed studies, we conclude that current management efforts do not improve all aspects of water quality. Additionally, there is no substantive evidence in the literature or PEIS that onsite mitigation by constructed channels or wetlands has replaced or will replace the lost ecosystem functions and biodiversity.

2. INTRODUCTION

The purpose of this report is to assess the state of the science on the environmental impacts of MTM-VF on streams in the Central Appalachian Coalfields.[1] The coalfields cover about 48,000 square kilometers (12 million acres) in West Virginia, Kentucky, Virginia and Tennessee, USA (see Figure 1) (U.S. EPA, 2003, 2005).

The Central Appalachian Coalfields have a long history of mining. Current mining methods, including MTM-VF, employ methods to control the acid mine drainages (AMD) that have been a historic and continuing source of water quality degradation. The purpose of this report is to evaluate evidence of the impacts of MTM-VF on headwater and downstream systems despite improvements in acidic discharges. It is prompted by EPA's re-examination of how best to implement environmental laws, especially the Clean Water Act (CWA), that are relevant to surface mining (see Section 2.2).

We evaluated six potential consequences of MTM-VF:

- Loss of headwater and forest resources (see Section 3)
- Impacts on water quality (see Section 4)
- Impacts from aquatic toxicity (see Section 5)
- Impacts on aquatic ecosystems (see Section 6)
- The cumulative impacts of multiple mining operations (see subsections of Sections 3, 4, and 6)
- Effectiveness of mining reclamation and mitigation (see Section 7)

We did not evaluate the impacts of MTM-VF on cultural or aesthetic resources.

We used two sources of information for our evaluation: (1) the peer-reviewed, published literature and (2) the PEIS and its associated appendices (U.S. EPA, 2003, 2005). Only a few peer-reviewed papers have studied water quality or stream ecosystems in headwaters directly affected by or downstream of MTM-VF in the Central Appalachian Coalfields (Appendix A). This report draws from these papers and from the relevant research findings of laboratory studies and observational studies from other locations and mining activities. We also discuss the findings published in the PEIS, which was published as two separate documents; the Draft, published in 2003, and the Final, published in 2005. The final PEIS included responses to comments on the draft and newer research results but did not include a revision of the original

[1]The derivation of the study boundary is described further in Chapter 4 of the PEIS (U.S. EPA, 2003, 2005).

material. When citing results from the many appendices of the PEIS, we specified the source to make it easier for readers to find the original material. Finally, authoritative text books were used as a source of background information and general scientific knowledge.

2.1. OPERATIONS USED IN MTM-VF

Mountaintop removal mining, like other surface mining practices, removes the soil and rock over a coal seam (i.e., the overburden) to expose the coal. It is distinct from other types of surface coal mining (e.g., contour mining) in its scale. This overview of the processes used in MTM-VF summarizes the description in the PEIS (U.S. EPA, 2003, 2005). The mountain or ridge top is prepared for mining by building access roads, clearing all trees and stockpiling topsoil for future use in reclamation. Then, explosives are used to blast the entire top of the mountain or ridge to expose and mine one or more coal seams (see Figure 2). As much as 300 vertical meters (1,000 feet) of overburden are removed.

The overburden removed during mountaintop mining cannot all safely be put back into place because of the overall volume of the material and because the volume increases when the rock is broken up. Some of the overburden is used to recontour the mine surface. The excess overburden is disposed of in constructed fills in valleys or hollows adjacent to the mined site. These fills bury the intermittent streams, springs and small perennial streams that comprise the headwaters of rivers.

Both water flow and sediment discharges are altered by MTM-VF (see Figure 3). The heavy equipment used to mine and move the overburden compacts the bare soils, forming a large, relatively impervious surface that increases surface runoff. On the mined site, surface runoff is diverted into ditches and sediment ponds, replacing natural subsurface flow paths. Water flows out of the ditches through notches, or is directed toward the valley fill. Depending on the construction and degree of compaction of the valley fill, the water then either percolates through porous fill material or flows through ditches and coarser rock drains within, under, or beside the fill. The effluent that emerges downstream of the ditches and below the downgradient edge (i.e., the toe) of the valley fill is discharged into constructed channels and then to ponds that are also used as treatment basins, for example, to settle solid particles, precipitate metals, or regulate pH.

After the coal is removed, the extraction area is graded and planted to control sediment runoff. The sediment retention pond may be eventually removed, and the stream channel is recreated under the footprint of the pond.

The coal is transported from the mine using trucks, conveyers or rail to a processing site, where it is washed prior to transport to market. The impacts of coal processing, slurry ponds and transport are not discussed in this report.

Mines can be as large as some cities (see Figure 4) and may use several different types of mining, including underground methods such as room and pillar or long-wall mining and surface methods such as contour, area and high-wall mining, in addition to mountaintop removal. Though these other forms of mining can also produce small fills, valley fills resulting from mountaintop removal are by far the largest. The active life of a mine increases with size; larger mines can be active between 10 and 15 years.

The density of all coal mining activity (surface and underground) can be quite high in some parts of the region (see Figure 5). Current statistics on the spatial extent of MTM-VF are unavailable. As of 2002, the footprint of surface mine permits was estimated at 1,634 km^2 (U.S. EPA, 2002) or about 3.3% of the land cover in the Central Appalachian Coalfields. As of 2001, permits for 6,697 valley fills were approved. Between removal with the mountain or burial under fill, over 1,900 km of stream were scheduled to be lost through these existing permits (U.S. EPA, 2002). The streams lost represent 2% of the streams in the study area, a length that is more than triple the length of the Potomac River. More current statistics were unavailable at the time this report was written, but both mine footprints and stream losses were projected to double by 2012 (U.S. EPA, 2002).

2.2. REGULATORY CONTEXT

MTM-VF are permitted by state and federal surface mining and environmental protection authorities. Individual mines are regulated under the Surface Mining Control and Reclamation Act (SMCRA) by the Office of Surface Mining (OSM) and by delegated States under OSM oversight. In addition, several specific sections of the CWA apply. These are implemented by the EPA, the U.S. Army Corps of Engineers (USACE) and individual states authorized to implement portions of the CWA. Although a complete listing and interpretation of the regulations that affect MTM-VF operations are beyond the scope of this paper, Appendix B provides a brief discussion of how water quality standards are implemented through the CWA in the context of MTM-VF.

Two CWA permits are relevant to MTM-VF. The USACE issues a permit pursuant to Section 404 of the CWA (33 U.S.C. § 1344) for the discharge of dredged and/or fill material. This permit includes the valley fill itself and the fill necessary to create a sediment pond below the valley fill. The second permit is issued by either the EPA or an authorized state pursuant to Section 402 of the CWA (33 U.S.C. § 1342). The Section 402 program is also known as the

National Pollutant Discharge Elimination System (NPDES). The NPDES permit includes the discharge from the sediment pond and any stormwater associated with the mining activity.

Both permitting programs prohibit activities or discharges that cause or contribute to violations of numeric or narrative state water quality criteria. While numeric criteria protect a water body from the effects of specific chemicals, narrative criteria protect a water body from the effects of pollutants that are not easily measured, or for pollutants that do not yet have numeric criteria, such as chemical mixtures, or suspended and bedded sediments. Examples of narrative standards that are particularly relevant to evaluating MTM-VF impacts include

- From West Virginia: *No significant adverse impact to the chemical, physical, hydraulic, or biological components of aquatic ecosystems shall be allowed* (WV § 47-2-3).
- From Kentucky: *Total dissolved solids or conductivity shall not be changed to the extent that the indigenous aquatic community is adversely affected* (401 KAR 10:031, Section 4(f)).

> "Adversely affect" or "adversely change" means to alter or change the community structure or function, to reduce the number or proportion of sensitive species, or to increase the number or proportion of pollution tolerant aquatic species so that aquatic life use support or aquatic habitat is impaired (401 KAR 10:001, Section 1(5)).

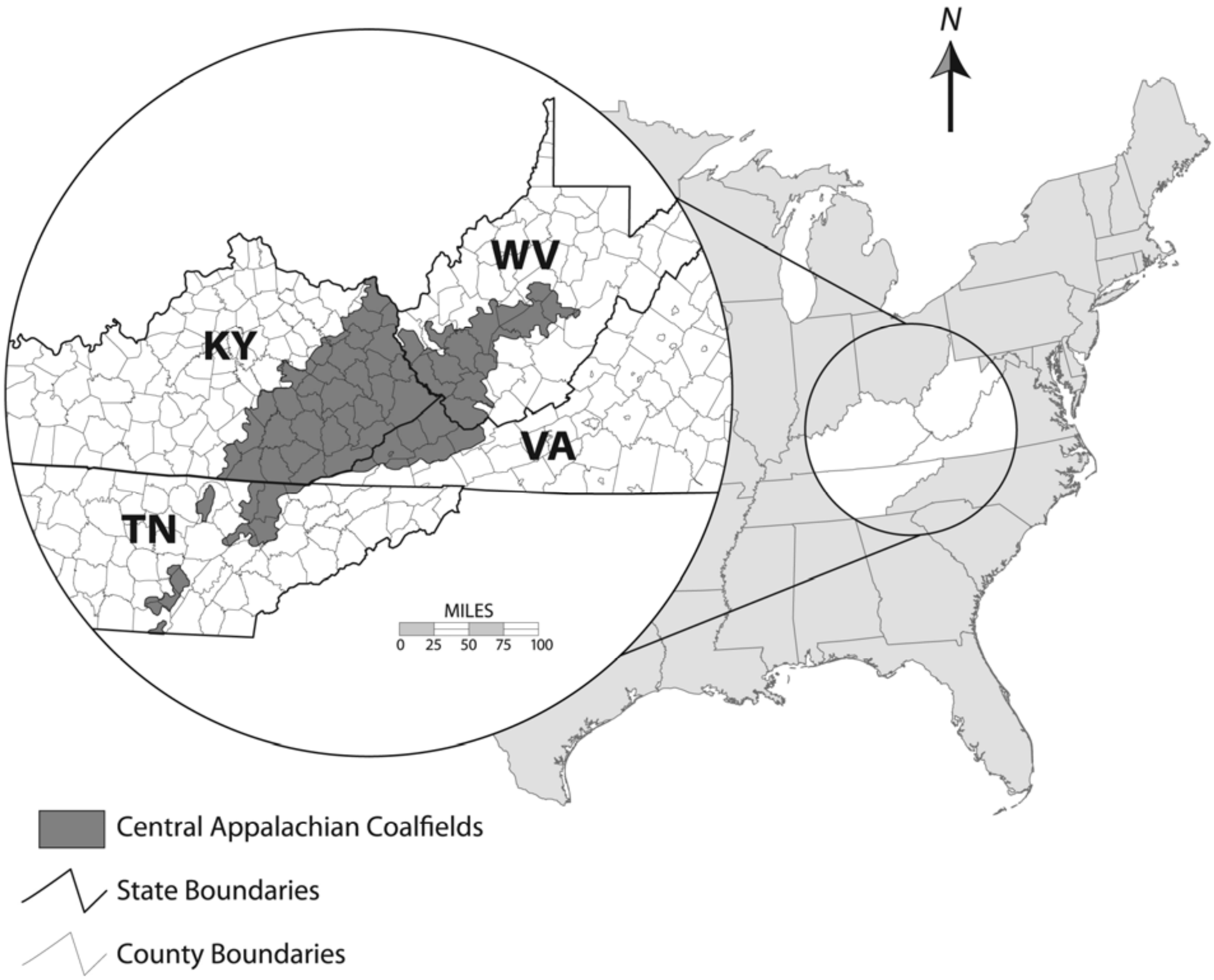

Figure 1. The Central Appalachian coalfields.

Source: EPA (U.S. EPA, 2003, 2005).

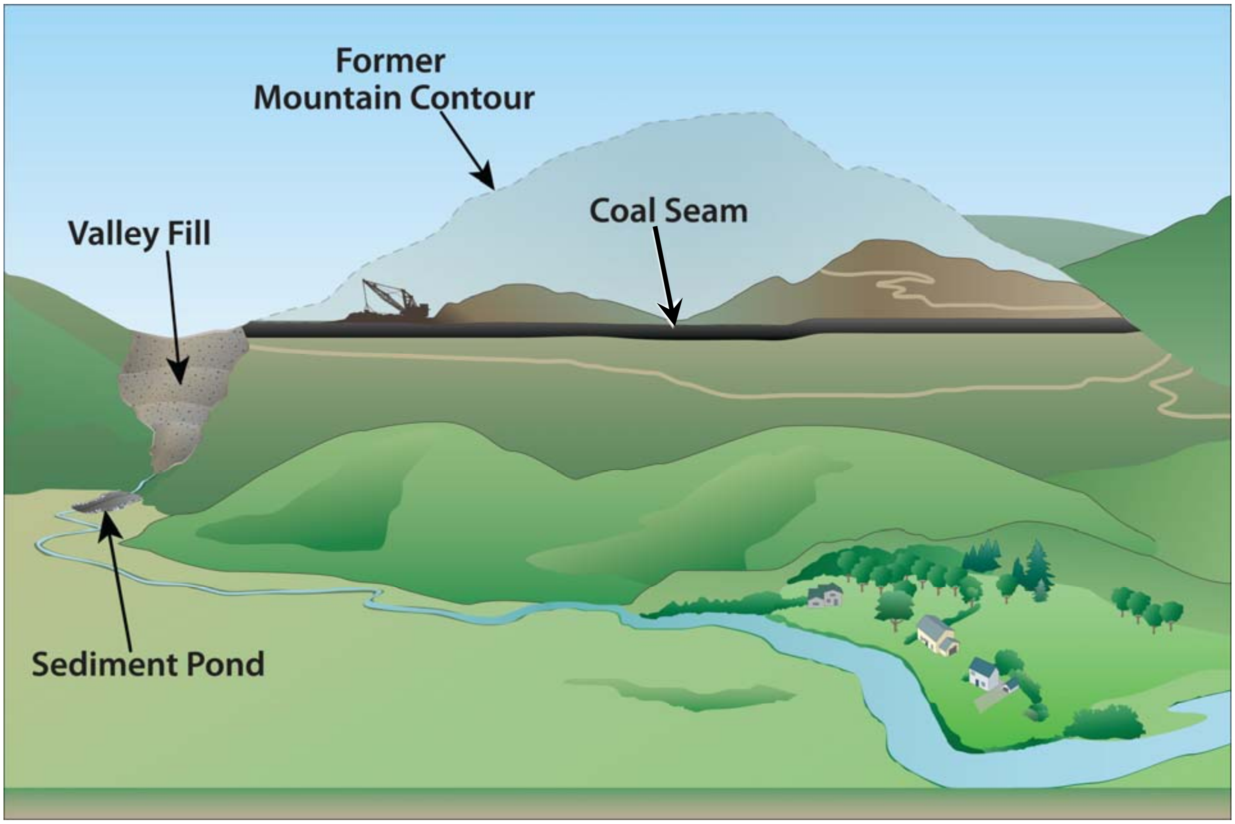

Figure 2. A watershed view of a mountaintop mine and valley fill (no consistent scale).

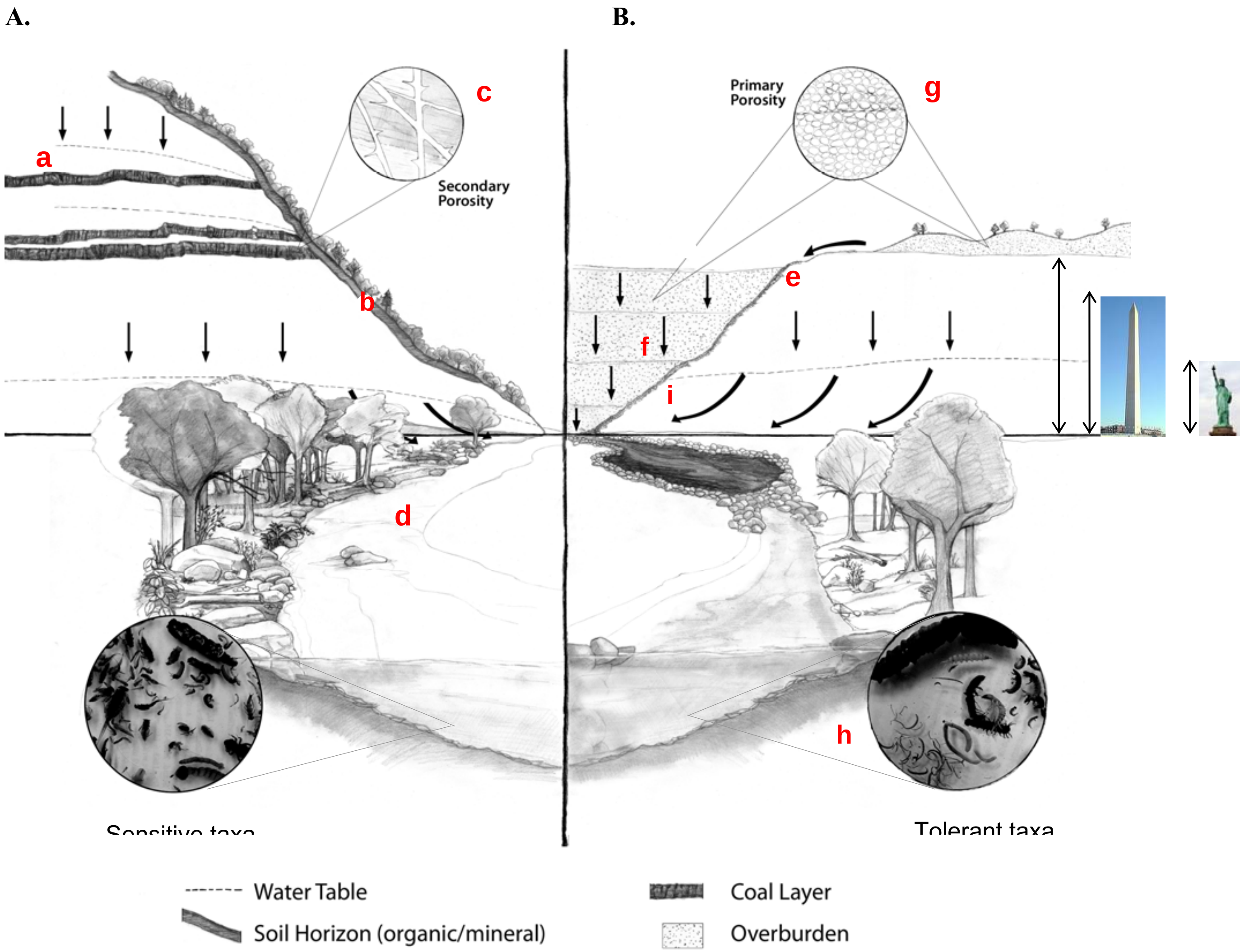

Figure 3. Small stream watershed before and after mountaintop mining and creation of a valley fill (simplified view, scales differ between upper and lower halves of diagram). Photographs of macroinvertebrates by Greg Pond.

A. Before mining. The figure on the left side of the diagram illustrates the natural topography, geologic strata and soil layers associated with small mountain streams in eastern coalfields. Stream valleys (natural depressions in the landscape that conduct channelized streamflow) are the most obvious topographic feature of the watershed. However, most of the water in small watersheds flows underground though a complex system of aquifers (a), soil layer interflows (b) and slow moving trickles through minute stress fractures in geologic strata of the parent mountain (c). Overland flow and subsurface flows (indicated by arrows) form channelized flows (d) that integrate features of the entire landscape, including riparian vegetation and diverse, instream biological communities.

B. After mining. On the right side, the same watershed is shown after the mountain rock layers have been removed, crushed and deposited in the stream valley. Flat surfaces of remaining rock layers are less permeable, producing higher surface runoff into a flood control channel (e) and valley fill (f, height is approximate). Infiltration though valley fills of water exposed to larger total surface area of porous unweathered rock (g) produces higher channelized flows and higher concentrations of dissolved ions and trace metals downstream, where biological communities shift towards tolerant taxa (h). Subsurface flowpaths in the intact geologic strata vary, depending on the types of rock in them, but water tables may 'back up' against the valley fill as shown here (i), increasing baseflows and exposure to valley fill materials.

B

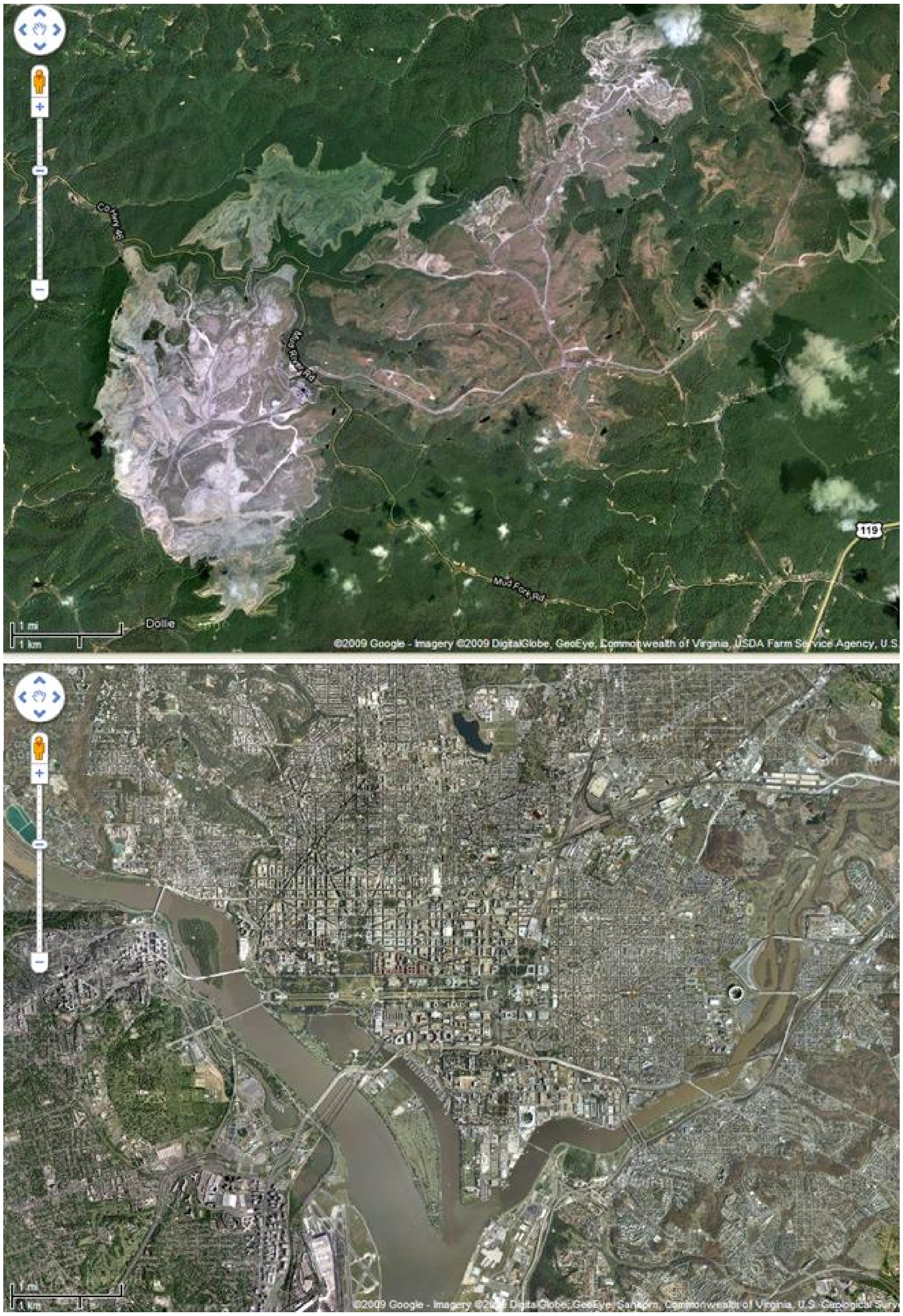

Figure 4. Satellite images of the 40-km^2 Hobet 21 mine (Boone County, WV) (Panel A), and the Washington DC area (Panel B), at the same scale.

Source: Google Maps (2009).

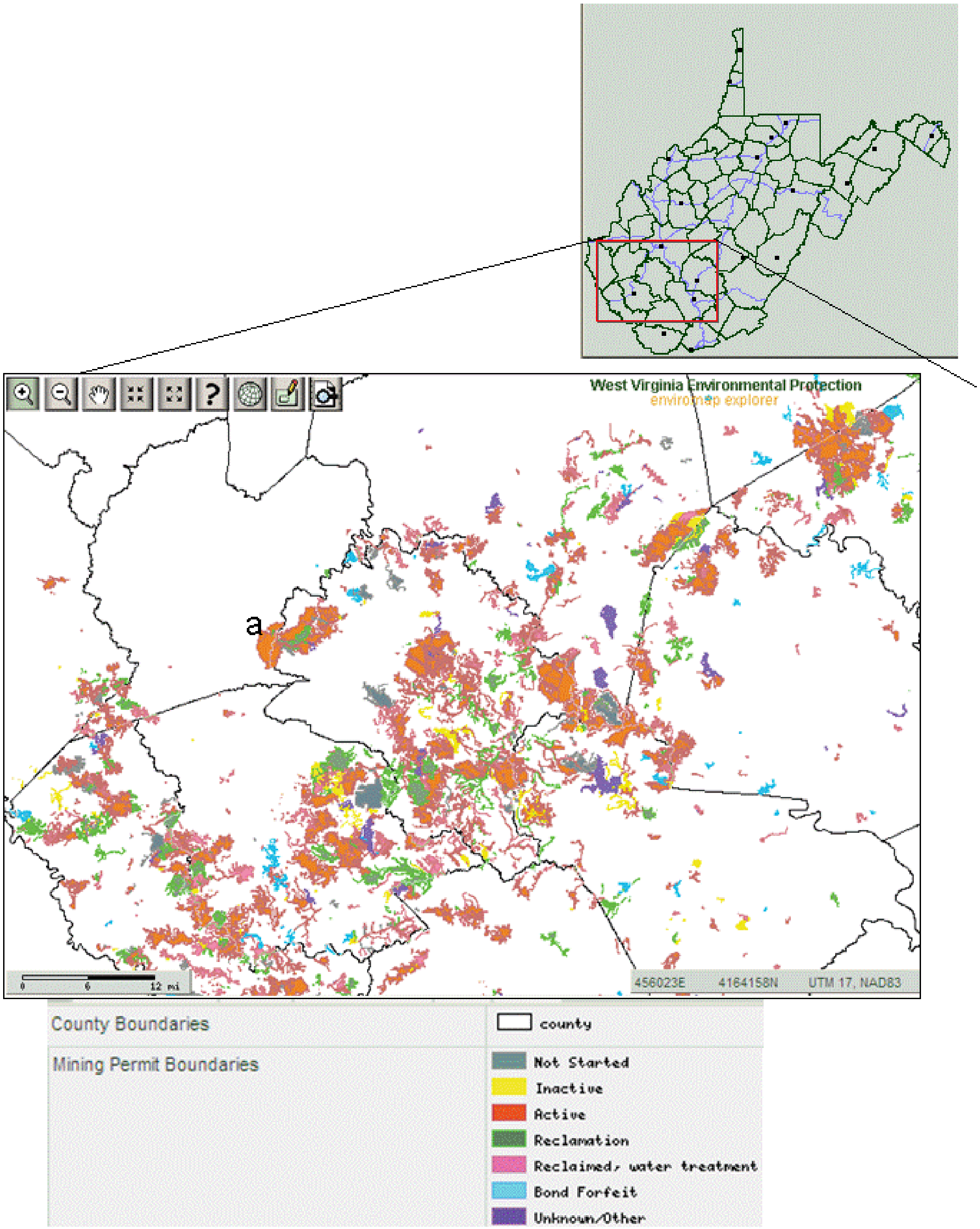

Figure 5. Permit boundaries for surface and underground mines in southwestern West Virginia. The Hobet 21 is shown in middle left near Point a.

Source: WV DEP (2009). Colors modified to improve legibility.

3. LOSS OF HEADWATER AND FOREST RESOURCES

Headwater streams dominate surface flows in the United States and comprise 70−80% of the total stream miles in the eastern coal mining states (Leopold, 1964; U.S. EPA, 2003, 2005). Headwater stream ecosystems occur on all mountains in the eastern coalfields and in all valleys that receive the excess overburden from mountaintop mining. Impacts include the loss of headwater streams and forests on the removed mountaintops; burial of streams in the actual footprint of the valley fills; and fragmentation of adjacent forests.

3.1. ESTIMATING EXTENT OF HEADWATER ECOSYSTEM LOSS

Estimating the extent of headwater ecosystems lost from MTM-VF begins with defining where streams begin. The term, *headwaters,* refers to the springs, seeps, creeks and seasonal and temporary flows that collectively form the origins of large river networks (see Figure 6). Headwater streams are formed by leakages that in turn form linkages. Headwaters form where groundwater breaks through (leaks) to the surface. At the surface, erosional processes create channels that form small (first order) streams that link groundwater, stormwater, upland and riparian ecosystems (see Figure 6) (Paybins, 2003; Freeman et al., 2007; Nadeau and Rains, 2007).

Headwater streams are classified as perennial, intermittent, or ephemeral based on the relative contributions of groundwater and stormwater inputs, the timing and duration of channelized (surface) flow, the drainage area, the channel's morphology and the underlying rock types (Hewlett, 1982). Perennial headwaters flow year-round and are predominantly groundwater-fed; intermittent streams flow seasonally (winter-spring) when groundwater levels are elevated; and ephemeral streams receive no groundwater input and flow only in response to precipitation events (e.g., rainfall, snowmelt) (Johnson et al., 2009).

OSM inventoried valley fills in the Central Appalachian Coalfields to estimate the number of stream miles lost to mountaintop mining and valley fills, based on a 0.12-km^2 (30-acre) minimum watershed size. This study found that in the 17-year period from 1985 to 2001, approximately 1,165 km (724 miles) of headwater streams were permanently buried under valley fills in West Virginia, Kentucky, Virginia, and Tennessee (U.S. EPA, 2003, 2005). In a cumulative impact study, the EPA (U.S. EPA, 2002) reassessed the number of stream miles lost by including streams that were lost to other mining activities (blasting, backfilling, etc.) in addition to valley fill footprints. In the revised estimate, 1,944 km (1,208 miles) of streams were approved to be lost due to valley fills and associated activities from 1992 to 2002 (U.S. EPA, 2003, 2005). This means that more than 2% of the total stream miles and 4% of first- and second-order stream miles in the PEIS study area were approved for permanent burial during this 10-year period.

Estimating headwater stream loss in terms of miles of stream impacted in watersheds above a size threshold is a useful beginning but does not address the loss of other headwater ecosystems. For example, the estimate does not include the springs, seeps and wet areas that may occur in places other than the stream channel and in watersheds less than 0.12 km^2 (30 acres) in size. Headwater stream burial estimated as the watershed area above the toe of valley fills on permits approved from 1985 through 2001 is shown in Table 1. The total area impacted shown here does not include valley fill permits approved prior to 1985 or after 2001.

In a study of 36 first-order streams for which valley fill permits were pending or approved, Paybins (2003) estimated that the median watershed area for intermittent flows was 0.1 km^2 (14.5 acres) and the median watershed size for perennial flows was 0.2 km^2 (40.8 acres). The average area of a valley fill shown in Table 1 is 0.3 km^2. These data suggest that intermittent and perennial streams are being buried by valley fills. The relationship of permits to valley fills is not one-to-one. Multiple permits may refer to the same valley fill, and large valley fills may cover more than one headwater basin (Paybins, 2003; see Figure 7).

3.2. LOSS OF HEADWATER ECOSYSTEM BIOTA

The biodiversity of the Central Appalachians is of national and even global significance. The Southern Appalachian and most of the Central Appalachian Mountains were a refuge for organisms during the last glacial maximum, which ended 10,000 years ago. The area includes one of the most prominent hot spots for high biodiversity measured as rarity-weighted species richness identified by NatureServe (see Figure 8). For example, nearly 10% of global salamander diversity is found within streams of the Southern Appalachian Mountains (Green and Pauley, 1987).

Evidence relevant to evaluating the loss of headwater ecosystem biota comes from surveys conducted as part of the PEIS and reports from headwaters in other temperate regions. We assume that most of the organisms inhabiting these systems are eliminated when the headwater is buried or blasted during the mining process.

Headwater habitats are spatially and temporally dynamic and support diverse biological communities (Gomi et al., 2002; Meyer et al., 2007; Clarke et al., 2008). Small—but biologically significant—differences in light, hydrology, water chemistry, substrate, sediments, food resources, gradient and precipitation across small streams within the same river network offer a wide variety of habitats and niches for aquatic and semiaquatic plants, animals and ecologically beneficial fungi and microbes (Meyer et al., 2007).

The effects of the loss of headwater biota on regional biodiversity would be expected to be most severe for taxa that occur only in headwater ecosystems, including intermittent streams (Morse et al., 1993; Morse et al., 1997; Hakala and Hartman, 2004). Intermittent streams can support diverse and abundant invertebrate assemblages (Feminella, 1996; Williams, 1996; Stout and Wallace, 2003). Some taxa inhabit only intermittent or only perennial streams, but many inhabit both types of streams (Feminella, 1996; Stout and Wallace, 2003; Hakala and Hartman, 2004). Populations of these taxa may be sustained by immigrants from nearby headwaters: intermittent streams can flow in all seasons in wet years; perennial streams may dry down seasonally during periods of drought and organisms may move freely between them. Stout and Wallace (2003) sampled 36 intermittent streams in West Virginia and Kentucky that were scheduled for burial by MTM-VF and collected approximately 73 genera and 41 families of aquatic invertebrates, many of which are found in perennial streams as well. In a stream with only subsurface flows over 70% of its length in summer, Collins et al. (2007) found that subsurface invertebrate community composition and water chemistry in the intermittent reaches was comparable to those in reaches with perennial surface flows.

In studies of two Appalachian headwater streams, more than 30 species of diatoms and more than 40 species of beneficial fungi were recorded (Gulis and Suberkropp, 2004; Greenwood and Rosemond, 2005). Diatoms and fungi are important food sources for fish and aquatic insects. In addition, fungi produce enzymes that are essential to the rapid decomposition of organic matter (e.g., wood and leaf litter). The breakdown of plant matter by fungi and other microbes make nutrients in difficult-to-digest vegetation accessible to fish and invertebrates (Gulis et al., 2006).

Headwater streams also support diverse and abundant assemblages of amphibians. Salamanders are the most common vertebrates in headwaters (Davic and Welsh, 2004). Many stream salamanders require headwater seeps and intermittent streams in forested habitats to maintain viable populations (Petranka, 1998; Davic and Welsh, 2004). High levels of genetic diversity in geographically distinct lineages of the spring-endemic Brownback Salamander (*Eurycea aquatica*, Plethodontidae) have recently been described in the Southern Appalachian Mountains (Timpe et al., 2009). Among the Appalachian plethodontids, species vary in their preferences for ephemeral, intermittent, or perennial headwaters to the extent that life stage and taxonomic information could be used to estimate hydroperiod at the collection sites (Johnson et al., 2009). Many amphibian species are most abundant in intermittent streams, perhaps because periodic drying offers freedom from predatory fish (Davic and Welsh, 2004).

Some species of salamanders split their lives between forests and headwaters and depend on a close connection in order to move between the two (Petranka, 1998). Cool, moist soils and large woody debris in the forested riparian zones of small streams provide suitable habitat for salamanders (Petranka, 1998). Forest clearing increases the dispersal distance between the two ecosystems and is expected to decrease the abundance of salamanders in small streams that remain at a site (Maggard and Kirk, 1998). Changes to the dendritic structure and terrestrial connectivity of natural headwater streams also decrease the number of salamander species found (Grant et al., 2009).

3.3. LOSS OF HEADWATER ECOSYSTEM FUNCTIONS

As with the loss of biota, we assume that all of the ecosystem functions performed by headwaters are lost when the headwater stream is buried or removed. These functions are lost not only to the headwater stream itself, but also to ecosystems downstream of the MTM-VF (discussed in Sections 5 and 6).

Because they are small, the contributions of headwater streams to ecosystem function at the watershed scale are often overlooked (Meyer and Wallace, 2001; Meyer et al., 2007). Although it is well known that a stream's ecological integrity depends on the functioning of its smallest tributaries, we do not know how to measure the incremental effects of stream loss on downstream functions.

Nutrient uptake and transformation occurs more rapidly in headwaters, where slower-moving waters have longer contact times with biologically and chemically reactive benthic substrates and hyporheic zones[2] of small, shallow channels (Alexander et al., 2000; Bernhardt et al., 2005). Peterson et al. (2001) estimated that 50−60% of the inorganic nitrogen entering a stream is retained or transformed in the headwaters, reducing downstream nutrient loads by half. This estimate is likely conservative because denitrification, a process that is known to occur in natural stream channels and riparian zones (Payne, 1981), eliminates nitrogen as N^2 gas and is not included in the estimate by Peterson et al. (2001). Riparian buffers have a central role in nitrogen removal, which is affected not only by buffer width and riparian vegetation, but also by soil type, subsurface hydrology, chemistry and interstitial biofilm communities in the riparian-hyporheic zone (Pusch et al., 1998; Mayer et al., 2007).

In addition to purifying water of nutrients, natural headwaters detoxify water of other contaminants including the metals copper (Cu), zinc (Zn), manganese (Mn), and iron (Fe) (Schorer and Symader, 1998). In contrast, outflows from filled headwaters typically are net exporters of toxicants to downstream segments (see Section 4). The loss of natural ecosystem

[2]Hyporheic zone: the subsurface ecotone below and adjacent to the stream channel, where surface water and ground water mix and exchange solutes. Much of the streamflow and biogeochemical processing in streams occur underground. The hyporheic zone also supports a rich variety of aquatic fauna (Boulton et al., 1998).

function and the export of toxicants act in combination to increase risk to water quality below MTM-VF.

In their natural state, forested headwaters typically transport little sediment or large woody debris by fluvial processes and act as sediment reservoirs for periods spanning decades to centuries (Benda et al., 2005). Substrate and organic debris dams slow the flow of water through headwaters, creating more contact time for processing organic matter, nutrients, and toxicants and regulating runoff in normal rain events. Recent evidence indicates that the number and distribution of small tributaries and the presence of forest cover, are the primary controls of runoff in high gradient watersheds (McGlynn et al., 2003; McGuire et al., 2005; Laudon et al., 2007).

Forested headwaters also receive and process large volumes of organic matter from upland and riparian vegetation (Wipfli et al., 2007). This terrestrial subsidy supports the biomass of animals, plants and fungi found in headwaters and downstream segments.

Headwaters and associated interstitial habitats provide refugia for macroinvertebrates during floods or spates and speed the recovery of aquatic communities when flow conditions improve (Angradi, 1997; Angradi et al., 2001). Headwaters also serve as nurseries and spawning grounds for amphibians and fish, including the brook trout (*Salvelinus fontinalis)*, the only trout native to West Virginia. Brook trout live and spawn in headwaters and often are the only fish present in Appalachian first-order and second-order streams (Hakala and Hartman, 2004). In a study of one West Virginia watershed, Petty et al. (2005) estimated that >80% of all brook trout spawning occurred in small streams (<3 km^2), including headwaters draining areas less than 0.25 km^2.

3.4. CUMULATIVE IMPACTS ON FOREST RESOURCES

The EPA's (2002) cumulative impact study evaluated ecological condition, biodiversity, forest loss and forest fragmentation in the Central Appalachian Coalfields. The 48,562 km^2 of the study area are dominated by 92% forest cover and contains roughly 95,000 km of streams, including 67,600 km (71%) of headwater streams. The total mining permit land area estimated from mine permit geographic information system (GIS) layers obtained from OSM was 1,634 km^2: 1,100 km^2 in Kentucky, 365 km^2 in West Virginia, 131 km^2 in Virginia and 38 km^2 in Tennessee.

3.4.1. Quantity of Forest Habitat Lost to Mountaintop Removal and Mining Infrastructure

Surface mining deforested 1,540 km^2 (3.4%) of the study area during the 10 years between 1992 and 2002. An estimated 5,700 km^2 (11.5%) of the PEIS study area was projected to be deforested by 2012, an area 1.4 times the size of the state of Rhode Island. This estimate does not recognize any reforestation efforts following mining and logging. The estimated cumulative habitat loss included a 3-fold increase in the area in former headwater stream watersheds classified in land use/land cover databases as "surface mining/quarries/gravel pits" (U.S. EPA, 2002).

3.4.2. Quality and Connectivity of Forest Habitat Lost

In its natural condition, the Appalachian landscape is dominated by interior forest. A decrease in forest cover followed by conversion to grasslands or other land cover has the potential to shift the fauna of the region from that found in intact, high elevation forests to one dominated by grassland and edge dwelling species.

Wickham et al. (2007) found that the pattern of deforestation from MTM-VF is destroying interior forests at a greater rate than would be expected from the overall rate of deforestation. Because of fragmentation, the area of interior forest lost was 1.75–5.0 times greater than the direct forest lost between 1992 and 2001. An increase in habitat fragmentation has the potential to isolate natural populations, reduce population sizes, reduce gene flow, increase the risk of extirpation or extinction of rare species and increase the rate of invasion by exotic species, especially plants (Harper et al., 2005; Ewers and Didham, 2006). Fragmentation of the terrestrial environment due to mining, projected from land cover data in the West Virginia Gap Analysis Program and the permit rates observed during the 10 years preceding the publication of the PEIS, indicates

- a 40% increase in the number of isolated forest habitat fragments,
- a 41% decrease in the average size of habitat fragments from 24.64 to 14.3 acres and,
- a 2.7% increase in the amount of edge habitat, caused by fragmentation of interior forests (U.S. EPA, 2002).

3.4.3. Riparian Habitat Lost

In the West Virginia portion of the study area, the projected loss of riparian habitat from MTM-VF is 30.72 km^2, 3.2% of the baseline. Approximately 42% of these projected losses occur in headwater (first- and second-order) streams (U.S. EPA, 2002).

Table 1. Watershed areas above the toe of valley fills approved from 1985 to 2001

Watershed area	Description
0.3 km^2 (71 acres)	Average size of watershed above valley fill toe
15.3 km^2 (3,774 acres)	Largest size watershed above valley fill toe
1,774.4 km^2 (438,472 acres)	Total watershed area impacted by valley fill construction

Source: EPA (U.S. EPA, 2003, 2005), Section III.

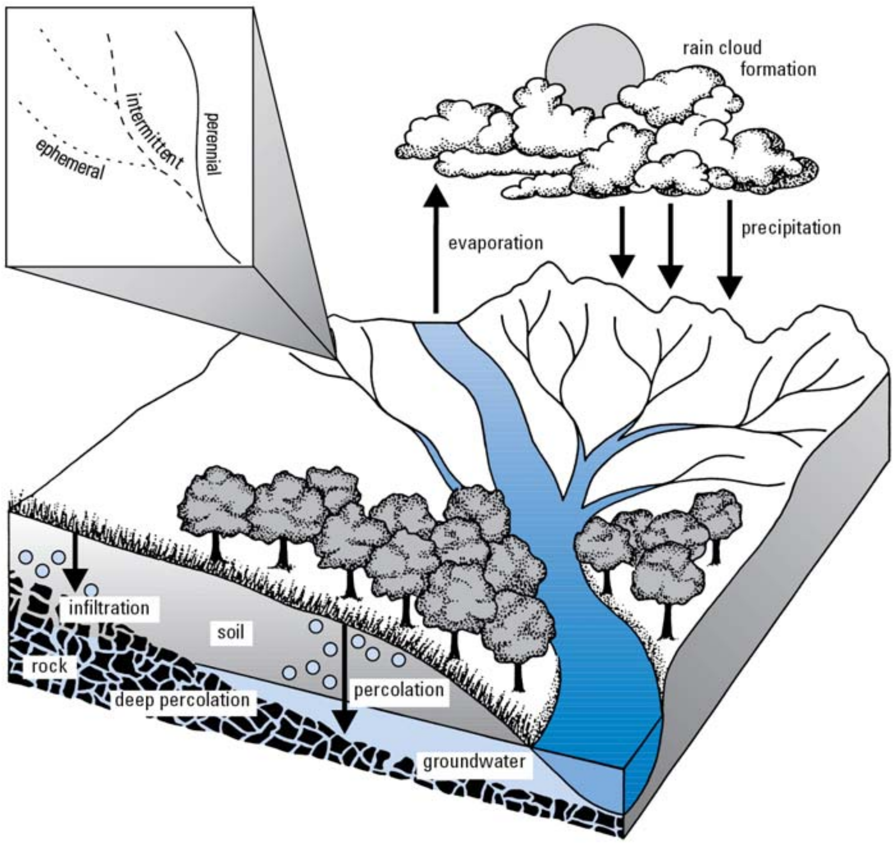

Figure 6. Headwater types: ephemeral, intermittent and perennial streams. Hillslopes erode to form ephemeral stream channels (dotted lines) that flow into intermittent (dashed lines) and perennial (solid line) streams.

Source: Jennings and Harmon (2002).

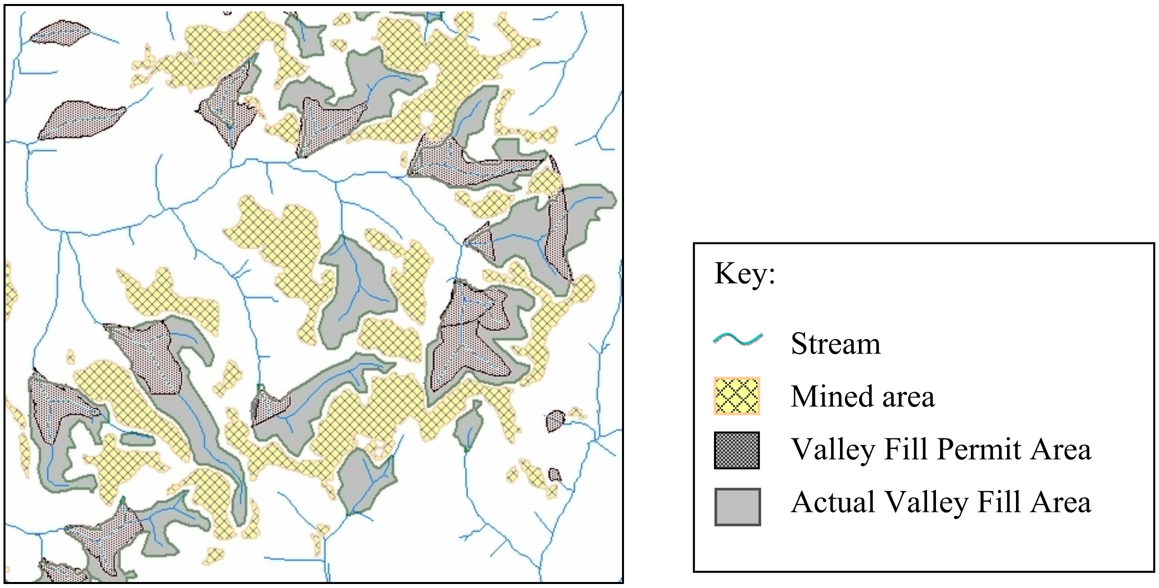

Figure 7. Map showing loss of headwater streams to MTM-VF. This diagram depicts the loss of stream miles and channel complexity that results from extensive mountaintop mining and valley filling. Blue lines inside valley fill areas represent buried streams. Note that the actual area of valley fill may exceed the area permitted for fill and that stream loss based on permit area may underestimate the full extent of buried streams.

Source: Modified from Shank (2004; Figure 12).

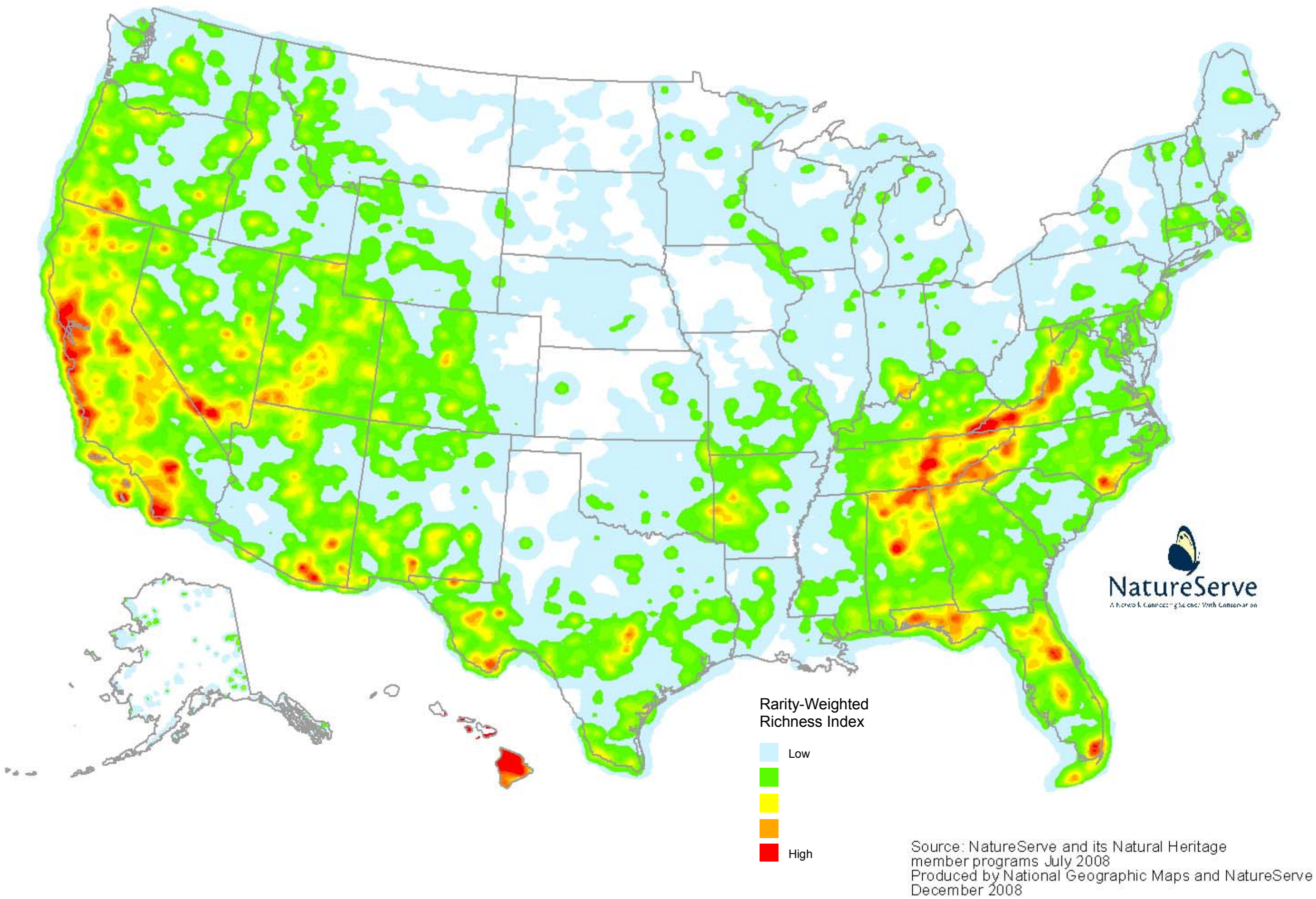

Figure 8. Hot spots of rarity-weighted species richness in the United States. The Central Appalachian Mountains, including the Central Appalachian Coalfields, have been identified as one of the most significant hot spots for biological diversity in the United States
Source: NatureServe and its Natural Heritage member programs, July 2008 (National Geographic Maps and NatureServe, 2008).

4. IMPACTS ON WATER QUALITY

In this section, we report the results of a number of studies that have assessed the changes in the physico-chemical attributes of streams downstream of MTM-VF. Although much of this information may also apply to constructed channels and other water-containing structures on the valley fills and the mined site, there are practically no data in the PEIS or the peer-reviewed literature on these constructed systems. The physico-chemical attributes we review below include alteration of stream flow, sedimentation of stream substrates, water chemistry, and sediment chemistry. Alterations of these attributes are the potential causes of the effects observed downstream of MTM-VF, which are described in Sections 5 and 6 of the report.

4.1. ALTERATION OF FLOW

Four factors may affect stream flow below valley fills. First, trees and other vegetation are removed from both the mined area and the area of the valley fill, and trees are generally slow to regrow on the mined area and valley fill. This reduces evapotranspiration rates from the watershed because transpiration is a function of the active vegetation (Dickens et al., 1989; Messinger, 2003). Second, the valley fill forms an unconsolidated aquifer in the watershed that stores a portion of any water that infiltrates into it (Dickens et al., 1989; Wunsch et al., 1999). This water comes from recharge along the periphery of the spoil body where surface-water drainage may be caught, from groundwater intercepted from adjacent bedrock aquifers, or from precipitation falling on the fill. Third, compaction of the fill surface by heavy equipment can reduce infiltration of precipitation and increase overland runoff (Negley and Eshleman, 2006). Fourth, when headwater streams are lost (see Section 3), attributes that influence surface flow (e.g., woody debris, surface water/ground water connections) are also lost.

Valley fills may act like a headwater aquifer and provide a more constant source of flow during the dry parts of the year. Comparing adjacent mined and unmined watersheds, monthly mean unit flow was relatively similar between the mined and unmined watersheds when soil and aquifer moisture levels were at their maximums in late winter and spring (February to May), but monthly mean flow in the mined watershed was greater than that in the unmined watershed during summer, autumn and early winter, when soil and aquifer moisture levels were reduced (Messinger and Paybins, 2003). Wiley et al. (2001) found the 90% duration flows[3] at sites below valley fills were 6 to 7 times greater than the 90% duration flows found at unmined sites. Moreover, daily streamflows from sites below valley fills were generally greater than those in

[3]The 90% duration flow is the streamflow (m^3/sec) equaled or exceeded at a site 90% of the time, a measure of the baseflow.

unmined watersheds during periods of low streamflow (Wiley et al., 2001). Green et al. (2000) observed that several of their unmined sites did not have surface flows during the summer and fall of a year when a drought occurred, but the streams below valley fills continued to have surface flows.

Storm intensity changes the relative effect of the valley fill on downstream flows. Intense storms can produce greater stormflows in watersheds with MTM-VF compared to unmined watersheds, but stormflows associated with precipitation from lower intensity storms may be ameliorated by valley fills. Messinger and Paybins (2003) found that a mined watershed had greater peak flows during severe storms than an unmined watershed. Unit peak flow[4] was greater in the mined watershed following summer thunderstorms when rainfall intensity exceeded 2.5 cm/hour (Messinger, 2003). In contrast, unit peak flow was lower in the mined watershed following low-intensity, long-duration rainfall events—particularly in the winter.

Wiley and Brogan (2003) found that peak discharges after an intense storm were greater downstream of valley fills than in unmined watersheds. Peak discharges were estimated by applying the slope-area method[5] (Benson and Dalrymple, 1967) to measurements of high water marks observed after flooding associated with a thunderstorm complex, which resulted in 7.6 to 15 cm of rainfall in southeastern West Virginia over a 5- to 6-hour period. Six sites were studied; three below valley fills and three in unmined watersheds. At two of the three sites downstream of valley fills, the estimated peak discharges were equivalent to floods that would naturally occur only once every 50 to >100 years. Peak discharges at the sites in unmined watersheds had less severe estimated flood recurrence intervals of 10 to 25 years (Wiley and Brogan, 2003). Estimates for the third site downstream of a valley fill were more difficult to interpret, as the peak discharge had an estimated flood recurrence interval of <2 years, much less severe than the other two mined sites. The differences might be due to differences in rainfall among the watersheds or differences in mine and valley fill attributes. Comparisons among sites assume that rainfall was similar among the watersheds, but there were differences among the sites from unmined watersheds. Moreover, thunderstorms can cause locally variable rainfall, particularly in mountainous terrains (Barros and Lettenmaier, 1994; Roe, 2005). Also, the third site was downstream from only a single reclaimed valley fill; there was no active surface mining in the basin, and the valley fill was larger than those of the other two sites, which both had active surface mining.

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[4]Unit peak flow is discharge per unit area of watershed, $m^3/sec/km^2$.
[5]With the slope-area method, the maximum flood height is estimated from the physical evidence left by the flooding, the high water marks. Then the cross-sectional area and wetted perimeter (i.e., the length of the part of the perimeter of the channel cross-section [stream bed and banks] below the water surface) of the stream channel is measured at that flood height. The slope of the stream bed is also measured, and Manning's *n*, an index of the roughness of the stream bed, is estimated. The peak discharge is then calculated using these variables.

4.2. CHANGES IN SEDIMENTATION

All valley fills are built with a sediment retention pond, which is intended to capture sand and finer-sized particles that are produced by the fragmentation of the overburden and may be washed downstream from the toe of the valley fill (U.S. EPA, 1979). Despite this, Wiley et al. (2001), using a modified Wolman (1954) pebble count for the bankfull channel,[6] found that the percentage of particles less than 2 mm (i.e., sand and fines) was elevated in stream reaches downstream from a valley fill (i.e., median = 60%, interquartile range = 56–65%) when compared to unmined streams (i.e., median = 24%, interquartile range = 15–34%).

Similarly, Green et al. (2000), using methods from EPA's Environmental Monitoring and Assessment Program for the wetted channel[7] (Kaufmann and Robison, 1998), found that mean substrate sizes were less in filled or filled/residential streams compared to unmined streams, while mean percentage of sand and fines was greater. However, mean substrate sizes were largest at sites described as being downstream of other types of mining without valley fills (i.e., generally older contour mines) (see Table 2).

Hartman et al. (2005) did not find any clear pattern of sediments in a study that compared pairs of sites using samples taken in December with a scoop sample separated with modified Wentworth sieves (McMahon et al., 1996) (see Table 3). In two cases, the proportions of sand and fines were similar; in the fourth case, it was greater in the filled site; and in the third case, it was greater in the reference site. However, there appears to have been a significant nonmining disturbance in this last control site, Big Buck Fork.

4.3. CHANGES IN CHEMICAL TRANSPORT AND BASIC WATER QUALITY PARAMETERS

4.3.1. pH, Matrix Ions and Metals

Almost invariably, coal mining exposes pyrite, a ferric sulfide mineral formed in association with coal (Caruccio et al., 1977; Altschuler et al., 1983; Casagrande, 1987; Younger, 2004). In the presence of water and oxygen (O_2), pyrite is oxidized (i.e., a reaction catalyzed by autotrophic bacteria) to form the strong acids characteristic of acid mine drainage (Stumm and Morgan, 1996):

$$Fe^{(2+)}S_2 + 3.75\ O_2 + 3.5\ H_20 \rightarrow Fe^{(3+)}(OH)_3 + 2\ SO_4^{2-} + 4\ H^+ \qquad \text{(Eq. 1)}$$

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[6]The bankfull channel is the entire channel, which is submerged at bankfull discharge—the point just before the stream flow begins to spread out onto the stream's flood plain at high flows. As a result, this approach measures some substrate that is dry during baseflow, which is when these channel characteristics are usually measured.

[7]The wetted channel is the portion of the channel that was submerged at the time these channel characteristics were measured.

However, in the presence of sufficient carbonate minerals, such as calcite ($CaCO_3$) and dolomite [$CaMg(CO_3)$], the acidity can be neutralized (Rose and Cravotta, 1998):

$$2\ CaCO_3 + 2\ H^+ \rightarrow 2\ Ca^{2+} + 2\ HCO_3^- \qquad \text{(Eq. 2)}$$

$$2\ CaMgCO_3 + 2\ H^+ \rightarrow Ca^{2+} + Mg^{2+} + 2\ HCO_3^- \qquad \text{(Eq. 3)}$$

The effluent waters from valley fills are generally not acidic and may be somewhat alkaline (Bryant et al., 2002; Merricks et al., 2007). The pH is generally 7.0 or greater (Bryant et al., 2002; see Tables 4-3, 4-4, 4-5). The alkaline pH has been attributed to exposure of the water to carbonate minerals within the valley fill that originate from fragmentation of the noncoal formations that form the overburden or are added during construction of the valley fill (Sobek et al., 1978; Banks, 1997; Skousen, 1997). Other methods that may moderate pH include physically isolating the pyritic materials within the mine or valley fill (Skousen et al., 2000) and treatment within the sediment retention pond.

Iron forms relatively insoluble compounds, such as $Fe(OH)_3$, under more alkaline conditions and may not be found in elevated concentrations in the effluent waters below valley fills (Bryant et al., 2002; see Table 4-3). However in some conditions, such as during higher flows, Fe can remain elevated (Hartman et al., 2005; see Table 4-5).

Most other metals, such as cadmium (Cd), chromium (Cr), Cu, lead (Pb), and Zn, coprecipitate with or sorb to the iron compounds (Kimball et al., 1995; Lee, 2002; Larsen and Mann, 2005) and were not found (in one study) at elevated concentrations in the effluent waters (Bryant et al., 2002; see Table 4). Exceptions to this are Mn and nickel (Ni), which may be elevated in the effluent waters below valley fills (Bryant et al., 2002; Hartman et al., 2005; see Tables 4 and 6). Mn may occur in association with siderite ($FeCO_3$) in shales within the overburden and is more soluble in the more alkaline waters (Larsen and Mann, 2005). Aluminum (Al) is found primarily associated with clay minerals in soils and is not soluble unless the pH is less than 4.9 (Nordstrom and Ball, 1986).

Sulfate (SO_4^{2-}), calcium (Ca^{2+} from calcite-type minerals), magnesium (Mg^{2+} from dolomite-type minerals) and bicarbonate (HCO_3^- from both calcite and dolomite), which are formed in the above reactions (see Eq. 1−3), are commonly present at elevated concentrations in the effluent waters (Bryant et al., 2002; Hartman et al., 2005). In addition, other water-soluble compounds within coal or overburden may be solubilized by the above reactions or just by the increased exposure to water in the fragmented overburden (Yudovich and Ketris, 2006; Vesper et al., 2008). These ions, including Se, K^+, Na^+, and Cl^- may occur at elevated concentrations in the effluent waters (see Tables 4 and 6). All these ions are components of the elevated specific conductivity, a measure of the stream's ability to conduct an electrical current, which reflects the concentration of dissolved ions in the water (measured in units of microSiemens per cm, μS/cm), and TDS observed in these waters (Green et al., 2000; Howard et al., 2001; Bryant et al., 2002; Bodkin, 2007; Merricks et al., 2007; Pond et al., 2008; see Tables 4–7). Hardness is another measure of these dissolved ions—particularly the divalent ones like Ca^{2+} and Mg^{2+}.

Most studies have not assessed the seasonal variability of water chemistry at these sites, but Green et al. (2000) present seasonal data for five consecutive seasons from 1999 to 2000. There appears to be little seasonal pattern to pH, but mean conductivities were greatest in all four watershed types during the summer sampling period, possibly because of seasonally reduced discharges (see Table 8). In particular, mean conductivity exceeded 1,000 μS/cm in streams in filled and filled/residential watersheds during the summer sampling period. In all seasons, conductivities at reference sites in unmined watersheds were an order of magnitude (10 times) lower than at sites in filled and filled/residential watersheds (see Table 8). Pond et al. (2008) observed conductivities up to 2,540 μS/cm in streams from mined watersheds.

4.3.2. Water Temperature

Valley fills reduce the annual variation in water temperature. Comparing mean daily water temperatures between an unnamed tributary of Ballard Fork near Mud, West Virginia; a stream downstream from a valley fill; and a reference site, Spring Branch near Mud, West Virginia; Wiley et al. (2001) found that mean stream temperatures were warmer downstream of the valley fill during the autumn, winter and spring, with the greatest difference being in February. In the summer, the mean stream temperatures downstream from the valley fill were cooler than those in the reference site. Moreover, the range of variation both annually and within different seasons was less downstream from the valley fill. The minimum and maximum temperatures downstream of the valley fill were 3.3ºC and 16.5ºC, respectively, while those in the reference stream were below 0ºC and 20.0ºC.

4.3.3. Nutrients

Bryant et al. (2002) found generally low median concentrations of nitrate (NO^3) plus nitrogen dioxide (NO^2) in streams from unmined watersheds and below valley fills, with some samples having concentrations less than the detection limit of 0.10 mg/L. However, the mean concentration of NO_3 plus NO_2 was slightly greater in the streams below valley fills (Pond et al., 2008), and a maximum concentration of 17 mg/L was observed. Bryant et al. (2002) speculated that this could be caused by use of nitrogen-containing explosives at these sites or by spreading nitrogen containing fertilizers during reclamation. Phosphorus (P) was not detected in any samples with a detection limit of 0.10 mg/L (Pond et al., 2008).

4.3.4. Dissolved Oxygen

In the studies that have measured dissolved oxygen (DO), concentrations in unmined streams and streams in either mined and valley fill streams have been reasonably high and similar among the different types of watersheds (see Tables 5 and 6; Green et al., 2000; Howard et al., 2001; Bryant et al., 2002; Hartman et al., 2005). Published concentrations range from 6.5 to 13.0 mg/L. However, no studies have looked at diurnal variation of dissolved oxygen in these streams.

4.4. CHANGES IN SEDIMENT CHEMISTRY

Data on sediment chemistry downstream of valley fills are limited to a study by Merricks et al. (2007). They sampled three to six stations at 100- to 150-m intervals in each of three streams downstream from sedimentation ponds below valley fills in West Virginia and a single reference site (see Table 9).

Sediment concentrations were generally greater at one stream, Lavender Fork, which was downstream from a reclaimed, 6-year old valley fill and that also had the greatest measured stream water conductivities. Sediment concentrations also generally decreased with increasing distance below the sedimentation ponds.

4.5. CUMULATIVE IMPACTS

In terms of downstream water chemistry, the primary cumulative impact of MTM-VF or at least the mining of coal by different methods in the region affected by MTM-VF has been elevated concentrations of SO_4^{2-} and conductivity. In larger streams of the Kanawha basin, Paybins et al. (2000) found that one-fourth of all water samples exceeded a SO_4^{2-} concentration of 250 mg/L and 70% of the water samples collected downstream of coal mines exceeded a regional background concentration of 21 mg/L that was calculated from data for basins with no history of coal mining. Moreover, the median concentration of SO_4^{2-} had increased by 1.6 times in these streams between 1980 and 1998, and conductivity had increased by 1.2 times (Paybins et al., 2000). SO_4^{2-} and some of the other ions contributing to conductivity are conservative ions in water, meaning that there are no chemical or biological processes removing ions from or adding ions to the waters. Therefore, any changes in SO_4^{2-} concentrations are the outcome of mixing of waters with differing SO_4^{2-} concentrations (Cooper, 2000). Therefore, the increased SO_4^{2-} and conductivity are associated with increased sources of water with elevated SO_4^{2-} and conductivity within the Kanawha basin. Because other land disturbances, such as residential development, are not origins of elevated SO_4^{2-} and conductivity, MTM-VF appear to be these sources.

Conversely, while total Fe, total Mn and total Al in many larger streams within mined basins exceeded regional background concentrations of 129 μg/L, 81 μg/L, and 23 μg/L, respectively, the median concentrations of total Fe and total Mn had decreased between 1980 and 1998 by approximately one-third and one-half, respectively, and pH had increased (Paybins et al., 2000). As discussed previously, these metals are not as soluble under more alkaline conditions, and their decrease may reflect the increase in pH associated with the increased sources of alkaline water within the Kanawha basin, the valley fills.

In the absence of other direct evidence on the cumulative effects of the changes in water chemistry associated with MTM-VF on downstream water quality, it should be noted that headwater streams, such as those affected by MTM-VF, have a large influence on downstream water quality. Alexander et al. (2007) found that 1st-order, headwater streams contributed 70% of the mean annual water volume in 2nd-order streams and 55% of the volume in higher-order rivers. For nitrogen, a nutrient that is not conservative like the ions associated with MTM-VF, these 1st-order streams contributed 65% of the flux in 2nd-order streams and 40% of the flux in higher-order rivers (Alexander et al., 2007).

In terms of sediment contaminants, Paybins et al. (2000) found significant concentrations of polycyclic aromatic hydrocarbons (PAH) at several stations within the Kanawha River basin (see Table 10). However, most of these PAHs appear to be constituents of particles of coal that occur in sediments because of the extensive coal mining and transport of coal in the region. Downing-Kunz et al. (2005) found sediment concentrations of coal ranging from 1 to 53 g/kg in streams draining more southern parts of the Central Appalachian Coalfields in Kentucky. PAHs are a natural component of coal (Chapman et al., 1996; Paybins et al., 2000), but these PAHs are unlikely to be bioavailable to benthic invertebrates or fish (Carlson et al., 1979; Ahrens and Morrissey, 2005; Yang et al., 2008). Arsenic (As) and metals were also detected in sediments (see Table 10) of the Kanawha River. However, the source of these sediment contaminants is less clear.

Table 2. Substrate measures

Substrate measure: mean (standard deviation)	Unmined (n = 9)	Filled (n = 15)	Filled/ residential (n = 6)	Other mined (n = 4)
Mean substrate size class (unitless)	3.7 (0.3)	3.5 (0.5)	3.6 (0.8)	4.0 (0.3)
Calculated mean substrate size (diameter in mm)	53	38	42	109
% ≤2 mm diameter (sand & fines)	16.9 (9.9)	20.7 (12.9)	29.7 (24.1)	8.0 (9.2)

Source: Green et al. (2000).

Table 3. Proportion of sediments that were sand and fines (mean [standard error]) in paired sites

Site names (reference/impaired)	Reference	Filled
W. Br. Atkins Creek/E. Br. Atkins Creek	0.35 (0.00)	0.46 (0.10)
Big Buck Fork/Hill Fork	0.78 (0.03)	0.50 (0.06)
Bend Branch/Rockhouse Creek	0.25 (0.07)	0.23 (0.02)
N. Br. Sugar Tree Creek/S. Br. Sugar Tree Creek	0.27 (0.02)	0.50 (0.04)

Source: Hartman et al. (2005).

Table 4. Water quality variables in unmined streams versus streams below valley fills

Variable	Unmined			Filled			Detection limit
	Median	Mean	Range	Median	Mean	Range	
SO_4*	12.6	16.0	11.0–21.6	523	696	155–1,520	5.0
Ca, total*	4.88	7.50	2.70–12.0	104	138	38.0–269	0.10
Mg, total*	4.10	4.30	2.30–7.00	86.7	122	28.0–248	0.50
Hardness*	29.1	42.0	17.0–72.0	617	801	225−1,620	3.3
TDS*	50.5	---	---	847	---	---	5.0
Mn, total	<0.005	0.034	<0.005– 0.083	0.044	0.14	0.009–9.0	0.010
Conductivity (µS/cm)*	66.4	62.0	34.0–133	585	1,020	159–2,540	---
HCO_3*	---	20.9	6.10–35.0	---	183	10.7–502	NA
Se, total*	<0.0015	<0.0015	<0.0015	0.012	0.011	<0.0015–0.037	0.003
Alkalinity	20.0	---	---	150	---	---	5.0
K, total*	1.58	1.60	1.30–2.00	8.07	9.90	3.00–19.0	0.75
Na, total*	1.43	2.40	0.70–5.50	4.46	12.6	2.60–39.0	0.50
Mn, dissolved	<0.005	0.021	<0.005–0.055	0.044	0.11	0.0065–0.85	0.01
Cl*	<2.5	2.8	<2.5–4.0	4.5	4.6	<2.5–11	5.0
Acidity	2.5	---	---	4.2	---	---	2.0
Ni, total	---	<0.010	<0.010	---	0.014	<0.010–0.059	0.02
NO_3/NO_2*	0.81	0.40	<0.10–0.90	0.95	3.4	0.80–17	0.10
pH (standard)*	6.8	7.1	6.1–8.3	7.8	7.9	6.3–8.9	---
Acidity, hot	<2.5	---	---	<2.5	---	---	5.0
Al, dissolved	<0.050	0.093	<0.050–0.19	<0.050	0.096	<0.050–0.27	0.10
Sb, total	<0.0025	---	---	<0.0025	---	---	0.005
As, total	<0.001	---	---	<0.001	---	---	0.002
Be, total	<0.0005	---	---	<0.0005	---	---	0.001
Cd, total	<0.0005	---	---	<0.0005	---	---	0.001

Table 4. Water quality variables in unmined streams versus streams below valley fills (continued)

Variable	Unmined			Filled			Detection limit
	Median	Mean	Range	Median	Mean	Range	
Cr, total	<0.0025	---	---	<0.0025	---	---	0.005
Co, total	<0.0025	---	---	<0.0025	---	---	0.005
Cu, total	<0.0025	0.0029	<0.0025–0.005	<0.0025	0.0026	<0.0025–0.0034	0.005
Pb, total	<0.001	0.0012	<0.0010–0.0021	<0.001	0.0012	<0.0010−0.0040	0.002
Hg, total	<0.0001	---	---	<0.0001	---	---	0.0002
Total organic carbon	1.4	---	---	1.4	---	---	1.0
P, total	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	0.10
Au, total	<0.005	---	---	<0.005	---	---	0.01
Th, total	<0.001	---	---	<0.001	---	---	0.002
V, total	<0.005	---	---	<0.005	---	---	0.01
Ba, total	0.029	0.040	0.015–0.072	0.025	0.041	0.022–0.068	0.020
Dissolved oxygen	13.6	---	---	11.0	---	---	---
Dissolved organic carbon	2.45	---	---	1.95	---	---	1.00
Total suspended solids	5.75	---	---	4.25	---	---	5.00
Fe, total	0.42	0.18	0.065–0.47	0.19	0.28	0.066–0.65	0.10
Fe, dissolved	0.22	0.074	<0.050–0.19	0.10	0.092	<0.050–0.28	0.10
Zn, total	0.0060	0.010	0.0033–0.023	<0.0025	0.0091	<0.0025–0.027	0.005
Al, total	0.15			<0.10			0.10

Units are mg/L, unless indicated otherwise. The table shows median, mean and range of concentrations for various water quality variables in unmined streams versus streams below valley fills from a data set reported on by Bryant et al. (2002) (median, 9 unmined sites and 21 filled sites, each sampled about six times from August 2000 to February 2001) and Pond et al. (2008) (mean and range for a subset that also had biological data, 7 unmined sites and 13 filled sites, except for pH and conductivity, which were measured at 10 unmined sites and 27 filled sites). If a concentration was less than the detection limit, the value is shown as <½ the detection limit. A "---" under median, mean, or range indicates that this variable was not reported in the indicated report. A "---" under detection limit indicates that there was no detection limit for that variable. A "NA" under detection limit indicates that no detection limit was reported for a variable only reported by Pond et al. (2008). An asterisk next to the variable name indicates that the mean concentration in streams below valley fills was statistically significantly greater than that in unmined streams at $p = 0.05$. A complete description of the analyses is found in Bryant et al. (2002).
Hg = mercury; Sb = antimony; Be = beryllium; Co = cobalt; Au = gold; Th = thorium; V = vanadium; Ba = barium.
Sources: Bryant et al. (2002)(median) and Pond et al. (2008).

Table 5. Water quality parameters for unmined or reference streams or streams downstream from mined, filled, or filled and residential watersheds in West Virginia

Variable	Green et al. (2000)				Merricks et al. (2007)		Hartman et al. (2005)	
	Unmined	Filled	Filled/ residential	Mined	Reference	Filled	Reference	Filled
Conductivity (µS/cm)	58–140 59 (38–178)	643–1,232 850 (159–2,500)	538–1,124 843 (155–1,532)	172–385 187 (90–618)	247 ± 87	923 ± 380– 2,720 ± 929	47.6 ± 2.4– 259.7 ± 30.6	502.0 ± 98.4– 1,479.0 ± 110.6
pH (standard)	7.1–7.5 7.5 (5.7–9.4)	7.1–7.9 7.7 (5.9–8.5)	7.1–8.3 8.0 (6.4–8.7)	6.7–8.4 7.4 (6.0–8.7)	7.2 ± 0.36	7.93 ± 0.18– 8.37 ± 0.47	6.5 ± 0.6– 7.0 ± 0.4	7.2 ± 0.6– 7.5 ± 1.0
Dissolved O_2 (mg/L)	6.5–13.3 10.9 (5.6–15.2)	7.5–13.0 10.0 (5.8–14.5)	8.5–14.0 9.4 (7.3–16.1)	8.7–12.7 10.2 (7.4–14.5)	---	---	8.5 ± 0.8– 13.4 ± 0.4	9.1 ± 1.0– 13.0 ± 0.6
Hardness (mg/L)	---	---	---	---	86 ± 20	544 ± 226– 1,904 ± 596	---	---

Sources: Green et al. (2000) (range of means among seasons, overall mean, overall range), Merricks et al. (2007) (range of means and standard deviations) and Hartman et al. (2005) (range of means and standard deviations).

Table 6. Alkalinity, pH and metals in control streams and streams downstream from filled watersheds in West Virginia

Parameter	Reference		Filled	
	Mean	**Range**	**Mean**	**Range**
Alkalinity*	12.8	0.400–46.8	163	16.2–319
pH (standard)	7.2	6.7–7.7	7.7	6.9–8.2
Na*	2.9	0.80–3.1	10	3.9–22
K*	3.3	1.5–5.1	10	1.8–14
Mg*	23	2.2–52	86	4.9–130
Ca*	37	2.6–67	130	5.9–200
Cu*	0.00080	0.00020–0.0019	0.0012	0.00050–0.0018
Ni*	0.0076	<0.00030–0.018	0.025	<0.00030–0.051
Zn	0.0027	0.0014–0.0047	0.0028	0.00090–0.0086
Mn*	0.019	0.0016–0.046	0.062	0.0020–0.17
Al	0.012	0.0090–0.019	0.019	0.00090–0.064
Fe*	0.016	0.0014–0.030	0.047	<0.00050–0.082

Units are mg/L, unless indicated otherwise. If the concentration was less than the detection limit, the value is shown as < the detection limit. An asterisk marks those measures where the fill streams were statistically significantly greater ($p < 0.05$) than the reference streams.

Source: Hartman et al. (2005).

Table 7. Dissolved oxygen, pH and conductivity for sites in eastern Kentucky

Parameter	Reference (n = 4)	Filled (n = 8)
Dissolved oxygen (mg/L)	9.1–9.6	8.4–9.7
pH (standard)	7.1–7.4	7.2–8.2
Conductivity (µS/cm)	30–66	420–1,690

Values are the range.

Source: Howard et al. (2001).

Table 8. Seasonal mean (standard deviation) of conductivity (μS/cm) for the four classes of streams

Season	Unmined	Filled	Filled/residential	Mined
Spring 1999	64 (19) n = 9	946 (614) n = 15	652 (237) n = 6	172 (90) n = 4
Summer 1999	140 (54) n = 2	1,232 (643) n = 15	1,124 (282) n = 6	385 (202) n = 3
Autumn 1999	91 (59) n = 2	958 (430) n = 14	984 (221) n = 6	260 n = 1
Winter 2000	73 (29) n = 9	836 (425) n = 14	844 (173) n = 6	254 (171) n = 3
Spring 2000	58 (28) n = 10	643 (382) n = 15	438 (249) n = 6	192 (155) n = 5

The number of sites (n) analyzed is also given.

Source: Green et al. (2000).

Table 9. Range of sediment concentrations of metals and arsenic (mg/kg) in streams downstream from the sedimentation ponds below valley fills in 2002 and 2004 and from a reference site in 2002

Metal or arsenic	Reference—2002 (n = 1)	Downstream from valley fill—2002 (n = 11)	Downstream from valley fill—2004 (n = 18)
Al	11	9–20	2–28
As	---	---	0.015–0.070
Cd	---	---	0.005–0.015
Cu	0.018	0.012–0.122	---
Fe	51	49–158	10–151
Hg	---	---	0.006–0.015
Mn	1.4	1.6–17	1.0–41
Se	---	---	0.001–0.011
Zn	---	---	0.1–2.5

The reference site was only sampled in 2002, and the analytes measured differed between the 2 years. The unmeasured analytes are indicated by ---.
Hg = mercury.

Source: Merricks et al. (2007).

Table 10. Polycyclic aromatic hydrocarbons, arsenic and metals detected in sediments of larger streams in the Kanawha Basin

Chemical and units of concentration	Number of detects/number of samples	Range of detections
benz[a]anthracene (µg/kg)	12/13	5–800
dibenz[a,h]anthracene (µg /kg)	4/13	40–200
2,6-dimethylnaphthalene (µg/kg)	10/13	50–500
fluoranthene (µg/kg)	13/13	30–1,100
fluorene (µg/kg)	7/13	60–300
naphthalene (µg/kg)	9/13	3–700
phenanthrene (µg/kg)	13/13	9–900
As (mg/kg)	13/13	4–20
Cr (mg/kg)	13/13	60–110
Pb (mg/kg)	13/13	20–50
Ni (mg/kg)	13/13	50–100
Zn (mg/kg)	13/13	200–600

Source: Paybins et al. (2000).

5. TOXICITY TESTS

In this section, we report on results of toxicity tests relevant to evaluating water quality downstream of MTM-VF. Toxicity tests expose organisms under laboratory conditions to ambient media (i.e., water or sediment samples), whole effluents, reconstituted effluents, or specific effluent constituents. Toxicity tests are valued because they can reflect the mixture as a whole, including antagonistic and synergistic effects. They also help distinguish the effects of water quality from other stressors (e.g., habitat quality, flow regime changes, temperature). Toxicity tests have been used as the basis for deriving water quality criteria and permitting industrial and waste water effluents.

The most common standard toxicity tests used to evaluate the effects of effluents measure the survival of the crustacean *Ceriodaphnia dubia* after 48 hours of exposure and the survival of fathead minnows (*Pimephales promelas*) after 96 hours of exposure. Both of these tests have limitations for evaluating MTM-VF effects: neither *Ceriodaphnia dubia* nor *Pimephales promelas* are native to the streams of the study area, and the standard test durations are much shorter than the exposures experienced by organisms downstream of MTM-VF operations. There are likely more sensitive responses than death. In particular, because ions are so influential in regulating membrane permeability during fertilization and egg development, effects on reproduction would be expected (Zotin, 1958; Ketola et al., 1988). Still, the standard survival tests provide a useful benchmark for understanding toxic potential. Other tests, which are more difficult and time consuming to run, can be used to extrapolate short-term tests on survival to longer-term exposures, sublethal responses and other species.

5.1. TOXICITY TESTS ON WATER COLUMN AND SEDIMENTS DOWNSTREAM OF MTM-VF

Only one study (Merricks et al., 2007) tested media downstream of MTM-VF within the study area. Water and sediment collected from some, but not all, sites downstream of valley fills produced significant toxicity in laboratory organisms.

Water was tested using *Ceriodaphnia dubia*. Results were reported as the percent dilution that killed one-half of the test organisms over 48 hours (48-hour LC50). Three streams were tested. The frequency of toxicity was highest in Lavender Fork; undiluted water from three of the eight sites killed 50% or more of the test organisms. Lavender Fork also had the highest specific conductivity levels; the undiluted water at the three toxic sites averaged 3,050, 2,497, and 2,657 µS/cm. Specific conductivity measurements were available for two of the five sites from Lavender Fork that did not result in 50% or greater mortality; specific conductivity measurements (2,720 and 2,667 μS/cm) were comparable to the toxic sites. Only 1 of 20 sites from the other two streams was sufficiently toxic to kill 50% or more of the test organisms. Specific conductivity measurements in these streams ranged from 923 to 1,643 μS/cm. There was no obvious relationship between toxicity and water column measurements of trace metals (e.g., Al, Fe, Mn, Zn, and Se).

Merricks et al. (2007) also conducted toxicity tests on sediments with another crustacean *Daphnia magna*. The organisms were exposed to sediments for 10 days; results were reported as percent survival and reproduction. Sediments from two of eight sites on Lavender Fork significantly reduced survival or reproduction of *Daphnia magna*. Sediments from 3 of 19 sites on the other two tested streams produced reduced survival or reproduction. Of the three streams, Lavender Fork generally had the highest concentrations of trace metals in sediments (i.e., Al, Fe, Cu, Cd, mercury (Hg), Se, As, Mn, and Zn). Concentrations of major ions or other chemicals were not measured. Because of the way the sediment chemistry results were grouped for summary, it is difficult to quantitatively relate them to the toxicity test results.

Asian clams (*Corbicula fluminea*) were deployed at monitoring stations (Merricks et al., 2007). Growth was significantly greater below the ponds and decreased downstream, indicating that the ponds increased the food available to the clams. Significant mortality was observed at 1 of 16 test sites. The authors attributed the mortality to Al and Cu, which had been detected in a previous, unpublished study at water concentrations of 223 and 7.6 μg/L, respectively.

5.2. TOXICITY TESTS ON WATER FROM OTHER ALKALINE COAL MINING EFFLUENTS

In a series of studies, Kennedy et al. tested the toxicity of a mining effluent from Ohio using *Ceriodaphnia dubia* and the mayfly *Isonychia bicolor* (Kennedy et al., 2003, 2004, 2005). The effluent originated from a surface mine, an underground coal mine and a preparation facility. Discharges from the underground mine and preparation facility were treated in a settling pond to neutralize pH and reduce Mn, resulting in an effluent with high SO_4^{2-}, Na^+, and Cl^- concentrations and a mean hardness of 770 mg/L as $CaCO_3$. Toxicity tests using *Ceriodaphnia dubia* were conducted following EPA protocols and used moderately hard reconstituted water (MHRW)[8] to dilute the effluent. Survival of *Ceriodaphnia dubia* in 48-hour tests significantly decreased relative to controls at a mean specific conductivity of 6,040 μS/cm (Kennedy et al., 2003). Decreased survival in 7-day tests was observed at a mean specific conductivity of 4,730 μS/cm. Decreased reproduction in 7-day tests was observed at a mean conductivity of

[8]Moderately hard reconstituted water (MHRW) was used as diluent in this study and many of the other studies discussed in section. MHRW has low chloride concentrations (mean of 1.9) and a Ca:Mg molar ratio of 0.88; hardness ranges from 80–100 mg/L as $CaCO_3$ (Smith et al., 1997).

3,254 μS/cm, about 1.9 times lower than the 48-hour results for survival (Kennedy et al., 2005). Tests on simulated effluent made using only the major ions (i.e., no heavy metals) agreed well with the whole effluent, providing evidence that the toxicity was caused by the ions, rather than an unmeasured toxicant (Kennedy et al., 2005).

The same field-collected effluent was tested with a nonstandard test species, the mayfly *Isonychia bicolor* (Kennedy et al., 2004) in 7-day tests. In these tests, water from an unpolluted reference stream was filtered and used as dilution water for the tests. Toxicity was greater at the warmer temperature tested (20°C vs. 15°C); those results are reported here. Survival of *Isonychia* significantly decreased relative to controls at specific conductivities of 1,562, 966, and 987 μS/cm for three tests. These conductivities are about 3 times lower than those that reduced *Ceriodaphnia* reproduction in 7-day tests using the same dilution water but a higher temperature of 25°C.

Chapman et al. (2000) tested a high sulfate alkaline coal mine effluent from Alaska in 10-day tests using the insect *Chironomus tentans*. No effects on chironomid survival were found, but dry weight was reduced approximately 45% in synthetic effluent (2,089 TDS/L). The researchers also tested the effects of synthetic effluent on rainbow trout using two exposures: eggs were exposed for 4 days starting immediately after fertilization, and swim-up fry were exposed for 7 days. No adverse effects were seen in embryo viability or fry survival in the highest synthetic effluent concentrations tested (2,080 TDS/L).

5.3. TOXICITY OF MAJOR IONS: K^+, HCO_3^-, Mg^{2+}, Cl^-, SO_4^{2-}, Na^+, Ca^{2+}

Laboratory studies that vary ion mixtures provide additional insight into which ions may be driving toxicity and how interactions may be producing observed effects. We report on the results of three of these study groups. Then, we compare the experimental results to ion concentrations reported downstream of MTM-VF operations to gauge whether ion concentrations would be expected to cause toxicity.

5.3.1. Mount et al., 1997

Mount et al. (1997) tested the acute toxicity of over 2,900 ion solutions using two crustacean species: *Ceriodaphnia dubia* and *Daphnia magna* and the fathead minnow (*Pimephales promelas*). *Ceriodaphnia* was the most sensitive of the three organisms. The toxicity of ion mixtures varied greatly with composition; total ion concentrations corresponding to acute LC50 for *Ceriodaphnia* ranged from 390 mg/L to over 5,610 mg/L. For *Pimephales promelas*, LC50 values ranged from 680 to 7,960 mg/L. The authors reported relative toxicity as $K^+ > HCO_3^- \approx Mg^{2+} > Cl^- > SO_4^{2-}$. They also developed regression models that could be used to predict the 48-hour acute toxicity of field-collected samples. In the models, the effects of the anions and cations were generally additive with two notable exceptions: Solutions with high concentrations of multiple cations had lower toxicity than expected based on concentration addition, and Na^+ and Ca^{2+} did not add any explanatory value after the other ions were included in the model.

The regression models have been used to predict the toxicity of several complex effluents. Tietge et al. (1997) used them to predict the acute toxicity of the ionic component of production waters to *Ceriodaphnia dubia, Daphnia magna* and *Pimephales promelas* (Tietge et al., 1997). Toxicity of the Ohio coal mine effluent (described above) to *Ceriodaphnia dubia* was less than expected based on the equations, although estimates were within a factor of 2 (Kennedy et al., 2005). Soucek (2007b) found that the model overestimated the toxicity of high hardness solutions to *Ceriodaphnia dubia* by a factor of about 5 (10% survival predicted vs. 50% survival observed).

5.3.2. Soucek (2007a, b); Soucek and Kennedy (2005)

Soucek (2007a, b) and Soucek and Kennedy (2005) conducted a series of 48-hour tests on SO_4^{2-} using MHRW dilution water and varying levels of other ions and hardness. At the highest hardness tested (600 mg/L), the 48-hour LC50 value for *Ceriodaphnia dubia* was 3,288 mg SO_4/L (Soucek and Kennedy, 2005). In all tests, the crustacean *Hyalella azteca* was the most sensitive test organism, followed by *Ceriodaphnia dubia*, the bivalve *Sphaertum simili* and the insect *Chironomus tentans*.[9] *Hyalella azteca* was particularly sensitive to SO_4 at low Cl^- concentrations. At Cl^- concentrations of 1.9 mg/L, *Hyalella azteca* was four times more sensitive to SO_4 than *Ceriodaphnia dubia* (Soucek, 2007a). Toxicity decreased as Ca increased relative to Mg concentrations (Soucek and Kennedy, 2005). Toxicity also decreased with increasing hardness, although the ameliorative effects of hardness appeared to level off above 500 mg/L hardness as $CaCO_3$.

In three-brood, 7-day tests on *Ceriodaphia dubia*, sublethal effects of SO_4 occurred at concentrations 2.5 times lower than those that reduced survival (Soucek, 2007a). The lowest concentration at which effects were significant compared with controls (the LOEC) was 899 mg SO_4/L for a reproductive endpoint (mean number of neonates per female) compared with 2,216 mg/L for percent survival. Other sublethal effects were investigated using 24-hour tests; significant declines in feeding rates and oxygen consumption were observed in *Ceriodaphia dubia* exposed to 1,000-mg SO_4/L.

[9] *Chironomus tentans* has since been renamed *Chironomus dilutus*

5.3.3. Meyer et al., 1985

Meyer et al. (1985) tested four salts using 48-hour tests on *Daphnia magna* and 96-hour tests on *Pimephales promelas.* High hardness dilution water was used (563 mg/L as $CaCO_3$). *Daphnia magna* was more sensitive to all of the salts than *Pimephales promelas*. The relative toxicity of the salts was $MgSO_4$ > NaCl > $NaNO_3$ > Na_2SO_4.[10] The LC50 values calculated for $MgSO_4$ were 4,300 mg/L and 7,900 mg/L for *Daphnia magna* and *Pimephales promelas*, respectively. All of these values are well above concentrations reported downstream of MTM-VF (see Table 4).

5.4. COMPARING TOXICITY TESTS ON MAJOR IONS TO OBSERVATIONS DOWNSTREAM OF MTM-VF

Applying the Mount et al. (1997) regression models to ion concentrations reported downstream of MTM-VF suggests that the ion mixture at some sites may reach acutely lethal levels. The models predict minimal mortality of *Ceriodaphnia dubia* (1%) at mean concentrations of each ion reported in Pond et al. (2008) (mean specific conductance of 1,023 μS/cm).[11] However, applying the assumption that ion concentrations are strongly correlated, we also calculated predictions using the maximum reported concentrations for each ion (maximum specific conductance of 2,540 μS/cm). More than 75% mortality is predicted at these maximum concentrations. The models predict minimal mortality (1%) for *Pimephales promelas* even at maximum concentrations.

Model predictions of toxicity are generally consistent with the observed *Ceriodaphnia dubia* toxicity test results reported by Merricks et al. (2007). Five sites tested by Merricks et al. (2007) had specific conductivity measurements comparable or greater than the maximum specific conductivity reported by Pond et al. (2008) (2,540 μS/cm). If the relative proportion of ions was the same in Merricks et al. (2007) as in Pond et al. (2008), we would expect these high conductivity sites to produce greater than 75% mortality. Three of these five sites exhibited 50% or greater mortality in 48-hour tests. Of the 11 sites with substantially lower specific

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[10] $MgSO_4$ = magnesium sulfate; NaCl = sodium chloride; $NaNO_3$ = sodium nitrate; Na_2SO_4 = sodium sulfate.

[11] For *Ceriodaphnia dubia*, proportion surviving (*P*) in 48-hour tests was calculated as logit $(P) = \ln[P/(1-P)] = 8.83 \times ((-0.0299 \times [K^+]) + (-0.00668 \times [Mg^{2+}]) + (-0.00813 \times [Cl^-]) + (-0.00439 \times [SO_4^{2-}]) + (-0.00775 \times [HCO_3^-]) + (-0.446 \times 2) + (0.00870 \times 2 \times [K^+]) + (0.00248 \times 2 \times [Cl^-]) + (0.00140 \times 2 \times [SO_4^{2-}]))$ (Mount et al., 1997). Concentrations are as reported in Pond et al. (2008) except for HCO_3^-. HCO_3^- concentrations were reported as $CaCO_3$ (Personal Communication from M.A. Passmore, U.S. EPA Region III, Wheeling, WV, 2009) and were converted to HCO_3^- concentrations by multiplying by 1.22.
For *Pimephales promelas,* proportion surviving (*P*) in 96-hour tests was calculated as logit $(P) = \ln[P/(1-P)] = 4.70 \times ((-0.00987 \times [K^+]) + (-0.00327 \times [Mg^{2+}]) + (-0.00120 \times [Cl^-]) + (-0.000750 \times [SO_4^{2-}]) + (-0.00443 \times [HCO_3^-]))$ (Mount et al., 1997). Concentrations are as reported in Pond et al. (2008) except for HCO_3^-. HCO_3^- concentrations were reported as $CaCO_3$ (Personal Communication from M.A. Passmore, U.S. EPA Region III, Wheeling, WV, 2009) and were converted to HCO_3^- concentrations by multiplying by 1.22.

conductivity readings (all less than 1,643 μS/cm), only 1 exhibited greater than 50% mortality in the toxicity tests.

The SO_4 48-hour LC50 for *Ceriodaphnia dubia* survival under high hardness conditions (3,288 mg SO_4/L) is twice the maximum concentration reported downstream of MTM-VF (see Table 4), suggesting that concentrations of SO_4 alone are unlikely to reach levels that are acutely toxic. However, SO_4 concentrations may be sufficiently elevated at some sites to cause sublethal effects in *Ceriodaphnia dubia.* The LOEC reported for reproductive effects in the 7-day tests (899 mg/L SO_4) falls between the median and maximum values reported in Table 4, suggesting that effects would be expected at many but not all sites downstream of MTM-VF. The higher hardness levels shown in Table 4 would be expected to ameliorate toxicity. However, bicarbonate and potassium concentrations are higher downstream of MTM-VF relative to the test conditions. Based on the Mount et al. (1997) tests described in Section 5.3.1, higher concentrations of bicarbonate and potassium would be expected to increase the toxicity of the mixture.

The toxicity tests on other alkaline mine effluents discussed in Section 5.3.1 suggest that effects to other organisms should be expected at concentrations below those that affect *Ceriodaphnia*. Tests using the mayfly *Isonychia bicolor* and the amphipod *Hyalella azteca* found effects on survival at concentrations 3–4 times lower than those affecting *Ceriodaphnia*. If effects on reproduction in these organisms are similarly more sensitive than survival, effects would be expected at most sites downstream of MTM-VF.

The relatively high sensitivity of mayflies to ions in alkaline mine effluent is consistent with relative sensitivity of mayflies to other salts. Mayflies were the most sensitive order of invertebrates tested in 72-hour laboratory studies of NaCl on South African invertebrate species (Kefford et al., 2004). In studies on metal salts in experimental streams and toxicity tests from the United States, the most sensitive invertebrates tend to be mayflies (Warnick and Bell, 1969; Clark and Clements, 2006). In studies on artificial seawater (dominated by NaCl) from Australia, the most sensitive species also were mayflies (Kefford et al., 2003).

Finally, there is some evidence that effects on survival of younger organisms may be observed at concentrations below those that affected the test organisms. The mayfly test reported by Kennedy employed older, larger instars of *Isonychia* (Kennedy et al., 2004). In tests with bicarbonate, 7-day-old *Hyalella azteca* were two times more sensitive than 14-day old organisms (Lasier et al., 1997). In studies on metal salts in experimental streams (Cu, Cd, and Zn), toxicity increased as organism size decreased (Kiffney and Clements, 1996).

5.5. TOXICITY OF TRACE METALS IN WATER

5.5.1. Selenium

Se is a metalloid element that is a micronutrient and, at higher exposures, a toxicant. Selenium from coal ash and coal mine wastes has resulted in elevated Se concentrations in surface waters and toxicity to aquatic organisms (Orr et al., 2005). Se is unusual in that its toxicity results from complex processes of transformation and bioaccumulation, analogous to mercury toxicity. Environmental exposures of animals are primarily dietary, and effects on sensitive early life stages are due primarily to maternal transfer. The current chronic AWQC for Se is 5.0 μg/L, and the median, mean and range of Se concentrations in streams draining valley fills are 12.5, 10.6, and <1.5–36.8 μg/L, respectively (Bryant et al., 2002; Pond et al., 2008). The chronic criterion is relevant because the discharge from mining operations is a chronic source. This section discusses effects of Se on aquatic invertebrates, fish and birds, emphasizing studies of waters receiving coal overburden leachates because the valley fills are filled with coal overburden.

5.5.1.1. *Invertebrates*

A review of the literature estimated that the range of thresholds for sublethal toxicity in aquatic invertebrate genera is 1–30 μg/L (DeBruyn and Chapman, 2007). A recent study showed that dietary selenium is bioaccumulated by the mayfly *Centroptilum triangulifer* and suggested that reproductive effects occur at aqueous exposures of 13.9 μg/L dissolved Se (Conley et al., 2009). These results are consistent with data from streams draining Canadian coal mines that found a >50% decline in the abundance of some taxa in the range of 5–100 μg/L (DeBruyn and Chapman, 2007). In outdoor artificial streams dosed with Se, isopods (*Caecidotea*) and oliogochaete worms (*Tubifex*) were severely reduced in abundance at 30 μg/L and statistically significantly reduced at 10 μg/L (Swift, 2002). However, the abundances of baetid mayfly nymphs (*Baetis, Callibaetis*), damselfly nymphs (*Enallagma*) and chironomid larvae were not statistically significantly reduced—even at 30 μg/L.

5.5.1.2. *Fish*

Numerous studies have shown severe effects of Se on fish reproduction in the field as well as in the laboratory, and effects on fish are the basis for the national criterion (U.S. EPA, 2004). Cutthroat trout embryos from a pond at a coal mine in British Columbia with 93 μg/L Se showed effects ranging from deformities of larvae to mortality (Rudolph et al., 2008). The probability of mortality was correlated with Se concentrations in the embryos. These trout are much less sensitive than other species such as bluegill sunfish. In the artificial stream study, bluegill sunfish exhibited mortality and characteristic skeletal deformities at all concentrations, including 2.5 μg/L, although the effects were not statistically significant at that lowest level (Swift, 2002). Creek chubs and blacknose dace from the Coal, Little Coal, Big Coal and Mud River watersheds in West Virginia contained Se from <0.48 to 6.89 mg/kg dry weight (Paybins et al., 2000). Fish from 3 of 22 of these streams had concentrations >5 mg/kg, putting them at "moderate hazard" for toxic effects based on the scale developed by Lemly (1993).

5.5.1.3. *Birds*

Se has caused reproductive failure and gross deformities in birds that forage in Se-contaminated waters, but their sensitivity is highly variable (Ohlendorf et al., 2003). Birds foraging in streams receiving leachate from coal mine overburden in the Elk River, British Columbia, watershed showed reproductive effects, but they were less severe than expected given the high Se concentrations (8.1–34.2 μg/L) (Harding et al., 2005). In particular, spotted sandpipers experienced a reduction in egg hatchability from 92% in reference streams to 78% in streams receiving overburden leachate. Spotted sandpipers forage in streams in the Appalachian Range, but the Louisiana waterthrush occurs more commonly in the area of concern and forages on aquatic invertebrates, so it would be similarly exposed. The authors suggest that the low level of effects relative to other Se-contaminated waters was due to low bioaccumulation, which was due to the low rates of biotransformation and uptake in those streams. Piscivorous birds (primarily Belted Kingfishers and Great Blue Herons) may be at risk from Se-contaminated fish. Creek chubs and blacknose dace from the Coal, Little Coal, Big Coal, and Mud River watersheds contained Se from <0.48 to 6.89 mg/kg dry weight (Paybins et al., 2000). The 10th percentile effective concentration for hatchability in dietary exposures of mallard ducks (a surrogate species for the piscivorous birds) to Se in dry diet was 4.87 mg/kg (Ohlendorf et al., 2003). Five of the 22 fish samples from 13 streams analyzed by Paybins et al. (2000) for Se from the Coal, Little Coal, Big Coal and Mud River watersheds exceeded that endpoint.

5.5.2. Manganese and Iron

Concentrations of Mn and Fe were higher at some sites downstream of valley fills compared with sites in unmined watersheds (see Table 6). In addition, Mn and Fe deposits have been observed on caddisflies downstream of MTM-VF (see Figure 9), suggesting that exposure is occurring under some circumstances (Pond, 2004).

Maximum concentrations of Mn reported downstream of MTM-VF are substantially lower than those associated with effects in the few available toxicity tests. Maximum concentrations of dissolved Mn reported in Pond et al. (2008) were 0.853 mg Mn/L. Tests using

Ceriodaphnia dubia in hard water (hardness = 184 mg/L) yielded a mean 48-hour LC50 of 15.2 mg Mn/L for *Ceriodaphnia dubia* and a 96-hour LC50 value for *Hyalella azteca* of 13.7 mg Mn/L (Lasier et al., 2000). In 7-day tests, *Ceriodaphnia dubia* reproduction (number of young per female) was inhibited 50% at mean concentrations of 11.5 mg Mn/L.

Pond et al. (2008) reports maximum concentrations of total and dissolved Fe as 650 μg/L and 281 μg/L, respectively. These concentrations are similar to the median tolerance limit concentration of 320 μg $FeSO_4$/L (water hardness = 48 mg/L) reported for *Ephemerella* sp. survival in a study conducted prior to standardized toxicity test protocols (Warnick and Bell, 1969). The concentrations are also greater than several of the family–level benchmarks for total Fe derived from field observations of benthic macroinvertebrates from West Virginia (see Table 11). Benchmark values (called field effect concentrations, FEC20s) corresponded to a 20% decline in the organism numbers compared with reference sites and were estimated from the 90th percentile quantile regression relationship between total Fe and numbers of organisms collected from different families. However, because the benchmark derivation did not control for stressors that covary with iron, the benchmarks may reflect the effects of other stressors in addition to iron.

5.6. TOXICITY OF TRACE METALS IN SEDIMENT

Only two studies measured concentrations of trace elements in sediments. Most concentrations were below available consensus-based screening levels (see Table 12). The consensus-based screening levels are based on analysis of paired sediment chemistry and toxicity test results from field studies and should be interpreted as concentrations at which effects in toxicity tests are frequently observed. Zinc and Ni concentrations in Kanawha Valley sediments exceed the probable effects levels and warrant further investigation. Toxicity of Zn and Ni is a function of particle size, organic carbon content, pH and acid volatile sulfides (Di Toro et al., 2001; Doig and Liber, 2006). It is difficult to interpret the observed concentrations without measurements of the factors that influence toxicity, or alternatively, pore-water concentrations.

Table 11. Field-based 20% effect concentrations (FEC20) (Linton et al., 2007)

Macroinvertebrate family	FEC20 (µg total iron/L)
Leptophlebiidae	210
Emphemerellidae	430
Philopotamidae	440
Psephenidae	480
Heptageniidae	660
Elmidae	1,130
Baetidae	1,480
Tipulidae	7,050

Table 12. Comparison of measured sediment concentrations with probable effects levels

Chemical	Concentration downstream of MTM-VF (mg/kg) (Merricks et al., 2007)[a]	Concentration in Kanawha Valley sediments (mg/kg) (Paybins et al., 2000)[b]	Consensus probable effects level (mg/kg) (MacDonald et al., 2000)[c]
Al	3–32		
As	0.015–0.070	4–20	33
Cd	0.005–0.045		4.98
Cr		60–110	111
Cu	0.019–0.122		149
Fe	<48.5–157.6		
Pb		20–50	128
Mn	1–41		
Hg	0.006–0.015		1.06
Ni		50–100	48.6
Se	0.1–2.5		
Zn	2.0–2.5	200–600	459

[a]Data from Table III and Figure 3 combined.
[b]Data from figures in appendix.
[c]We note that the concentrations reported in Merricks et al. (2007) are substantially lower than ranges of values reported in Paybins et al. (2000) or used to develop the PECs (e.g., see Smith et al., 1996) suggesting that any comparisons should be made with caution.
Blank cells indicate that the metal was not measured, or there is no probable effects level available.

Figure 9. Mn (black) and Fe (orange) deposits on a caddisfly collected downstream of a mountaintop mine and valley fill.

Source: Pond (2004).

6. IMPACTS ON AQUATIC ECOSYSTEMS

In this section, we report on changes in stream community composition associated with MTM-VF. In contrast to the toxicity tests discussed in Section 5, field studies are our primary resource for this section because they directly consider both the exposures and biota of interest. Macroinvertebrate and fish assessments indicate degraded biological condition downstream of MTM-VF.

6.1. EFFECTS ON BIOLOGICAL COMPOSITION

Mountaintop mining and associated valley fill in a watershed is associated with degraded community composition in downstream habitats. Though there are relatively few studies on the direct ecological effects of mountaintop mining, the findings are unambiguous (Howard et al., 2001; Stauffer and Ferreri, 2002; Fulk et al., 2003; Hartman et al., 2005; Merricks et al., 2007; Pond et al., 2008). Across all the relevant studies reviewed, mayfly, i.e., insect order Ephemeroptera, populations were consistently lower in streams draining watersheds with MTM-VF than in streams draining watersheds with intact forest. Associated with the extirpation of mayfly species, biological assessment metrics indicate degraded conditions immediately downstream of MTM-VFs.

6.1.1. Benthic Macroinvertebrates

6.1.1.1. *Benthic Macroinvertebrate Indices*

All surveys that used multimetric and aggregate taxonomic indices observed degraded biological conditions in streams affected by mining and valley fills (see Table 13). Fulk et al. (2003) used the West Virginia Stream Condition Index (WV SCI) to analyze benthic macroinvertebrate data from 34 streams in West Virginia. The index is composed of several metrics that are responsive to environmental and chemical stress, e.g., EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa, total taxa and percent EPT were expected to decrease with increasing stress and percent Chironomidae; Hilsenhoff biotic index (HBI) and percent of the top two dominant taxa were expected to increase with increasing stress. Four classes of streams were compared: (1) no mountaintop mining upstream (n = 9), (2) upstream valley fills (n = 15), (3) mountaintop mining in watershed (n = 4) and (4) upstream valley fill and residential development in the watershed (n = 6). Fulk et al. (2003) observed that benthic macroinvertebrate indices were lower in streams with upstream valley fills. With the exception of the fall of 2000, the macroinvertebrate index showed significant differences among stream classes, and these differences were caused by fewer total taxa and fewer EPT taxa in streams with valley fill.

While unmined sites were typically classified as "very good" streams with upstream valley fills had WV SCI scores that ranged from "fair" to "good", indicating that stream sites with valley fill were degraded compared to unmined sites. Similar results were observed in an assessment of the biological condition of streams in Kentucky. Howard et al. (2001) calculated the Kentucky Macroinvertebrate Bioassessment Index (MBI) (Pond and MacMurray, 2002), which includes four components of macroinvertebrate community condition, and showed that streams with mining activity in the watershed in Kentucky had lower MBI ranks than streams in watersheds without mining (mined streams had a rank of poor, and reference streams were good). Merricks et al. (2007) observed a 1.91 HBI score in a reference stream, indicative of excellent water quality, and HBI values ranging between 5.64 and 5.7, indicative of fair water quality, below valley fills and ponds in three streams. Finally, in a comparison of streams with and without mining in the watershed, Pond et al. (2008) observed that streams below fills had a significantly lower macroinvertebrate biotic index score than those without fills using both a genus-level index of most probable steam status (GLIMPSS, 2.4 vs. 4.5, respectively) and a family-level biotic index (WV SCI, 3.4 vs. 4.3, respectively).

Most field studies reported a reduction in commonly used measures of sensitive macroinvertebrates, the aggregated EPT metrics, at sites downstream of MTM-VF. EPT taxonomic richness was lower in one study (EPT generic richness of 17.9 at unmined sites and 8.9 at filled sites, Pond et al. 2008) and showed no significant difference in another (Merricks et al., 2007). Hartman et al. (2005) observed no difference in EPT richness between mined and unmined streams. An EPT index was lower in streams in mined watersheds compared to measures in streams in watersheds without mining activity (an average of 8.9 in mined sites and 21 in reference sites) (Howard et al., 2001). And the percentage of the community comprised of EPT taxa was lower at sites downstream of MTM-VF (Merricks et al., 2007; Pond et al., 2008). The mixed effects of mining on EPT aggregate measures likely reflect legacy land-use differences, differences in location of sample sites (e.g., sampling close to a pond) and taxonomic shifts within insect orders.

6.1.1.2. *Benthic Macroinvertebrate Diversity*

In most cases, lower taxonomic diversity was observed at sites downstream of MTM-VF. A pattern of lower macroinvertebrate richness in streams with mining in the watershed was found in Kentucky (mean of 31 at mined sites and 43 at reference sites, Howard et al., 2001) and in West Virginia (mean generic richness of 21.7 at mined sites and 31.9 at unmined sites, Pond et al., 2008). In contrast, Merricks et al. (2007) found no significant difference in taxonomic richness between filled streams and a stream without fill in the watershed.

6.1.1.3. *Benthic Macroinvertebrate Density*

No difference was found in the total density of macroinvertebrates between streams with valley fill and reference streams (Hartman et al., 2005).

6.1.1.4. *Benthic Macroinvertebrate Functional Groups*

MTM-VF were associated with changes in the functional composition of macroinvertebrate communities. Typically, macroinvertebrate communities in headwater streams are dominated by shredders, which feed on leaf detritus (e.g., Vannote et al., 1980). In the case of mining activities, shredder density metrics (Hartman et al., 2005) and proportion of the community (3% in streams with mining and 50% in a reference stream, Merricks et al., 2007) were lower in streams below fills. Other changes include lower percentage of the community as clingers (i.e., organisms that cling to rocks in riffles) in mined watersheds than in watersheds without mining (Howard et al., 2001). Also, a scraper (i.e., organisms that feed on attached algae) density metric was lower in filled streams than it was in streams without valley fill in the watershed (Hartman et al., 2005). The percentage of the community as collector-filtering macroinvertebrates (i.e., organisms that feed on floating algae) was higher in streams below fills (Merricks et al., 2007), but a collector density metric showed no difference between streams below fills and reference streams in another study (Hartman et al., 2005).

6.1.1.5. *Benthic Macroinvertebrate Taxa*

Specific changes in macroinvertebrate taxonomic composition are described below.

6.1.1.5.1. *Coleoptera.*

Only one study included coleopteran populations in their assessment found that a density metric of Coleoptera was lower in streams below valley fill than in streams without valley fills in the watershed (Hartman et al., 2005).

6.1.1.5.2. *Diptera.*

The effect of MTM-VF on Diptera population characteristics was mixed. In some cases, there were no observed effects of fills or mining on the watershed. For example, perhaps owing to moderate degradation in the reference sites, density metrics for Diptera and Chironomidae, a family within the insect order Diptera, showed no difference between streams downstream of valley fills and those without (Hartman et al., 2005). Merricks et al. (2007) had similar findings, where the percentage of the community comprised of Chironomidae showed no difference between sites downstream of valley fills and a stream without fills (Merricks et al., 2007). In another study, the percent Chironomidae was lower in streams with mining in the watershed than in streams with no mining (27% in mined and 13% in unmined streams, Pond et al., 2008). A combined measure of the percent Chironomidae and Oligochaeta was higher in streams in mined watersheds compared to streams in watersheds without mining (63% in mined and 3% in reference streams, Howard et al., 2001). The family Chironomidae includes both tolerant and intolerant taxa, which may account for the equivocal results.

6.1.1.5.3. *Ephemeroptera.*

Ephemeroptera population characteristics showed the most definitive changes associated with mining activities, where they were consistently lower in streams affected by MTM-VF. Ephemeroptera density metrics were lower in sites downstream of valley fills than in streams without fill (Hartman et al., 2005). The proportion of the community as Ephemeroptera was lower in impacted streams. Howard et al. (2001) found an average of 1% in streams with mountaintop mining in the watershed and 55% in reference streams. Two additional studies report similar observations of proportion (3% in streams with mountaintop mining in the watershed and 17% in reference streams [Merricks et al., 2007]; 7% in streams with mountaintop mining in the watershed and 45% in streams with no mining [Pond et al., 2008]). Likewise, Ephemeroptera richness was significantly lower in mine-impacted streams (Merricks et al., 2007, Pond et al., 2008).

6.1.1.5.4. *Odonata.*

An Odonata density metric was significantly lower at sites downstream of valley fills than it was in streams without upstream valley fills (Hartman et al., 2005).

6.1.1.5.5. *Plecoptera.*

MTM-VF had mixed effects on Plecoptera populations. In one case, richness was lower in streams with mining in the watershed (2.7 genera) than in streams without mining (6 genera) (Pond et al., 2008). But in another case, there was no significant difference in Plecoptera richness between sites downstream of valley fills compared to those without upstream fill (Merricks et al., 2007). A similar discrepancy was found with percent Plecoptera, where Merricks et al. (2007) found lower percentages in sites downstream of valley fills (4.5% in mined streams and 52% at a reference site), whereas Pond et al. (2008) did not detect a difference between streams with mountaintop mining in the watershed and streams with no mining in the watershed. No difference was observed in a Plecoptera density metric between streams with and without valley fills in Hartman et al. (2005).

6.1.1.5.6. *Trichoptera.*

MTM-VF had mixed effects on Trichoptera populations in streams. When the area below the pond was sampled, the proportion of the macroinvertebrate community that was in the order Trichoptera was higher in streams with mining in the watershed compared to streams without mining (20% in mined and 3.7% in reference watersheds, Merricks et al. 2007). Another study found no difference among streams downstream of fills and those without fills (Hartman et al., 2005).

6.1.1.5.7. *Noninsect Benthic Macroinvertebrates.*

A density metric of noninsect macroinvertebrates was significantly lower in at sites downstream of valley fills than in streams without fills (Hartman et al., 2005).

6.1.2. Fish

Fish community attributes, e.g., species richness, are widely used to evaluate stream condition (Karr, 1981; Angermeier et al., 2000). Species richness and the number of benthic fish species were consistently associated with site quality in Mid-Atlantic Highland streams (Angermeier et al., 2000), where both attributes declined with increasing degradation (Barbour et al., 1999). Mountaintop mining for coal and creation of valley fills has had a harmful effect on the composition of stream fish communities. Comparison of streams without mining in the watershed and sites downstream of valley fills in Kentucky (five unmined sites and seven filled sites) and West Virginia (four unmined sites and two mined sites) indicate that streams affected by mining had significantly fewer total fish species and fewer benthic fish species than streams without mining in the same areas (Stauffer and Ferreri, 2002). A similar pattern of fewer taxa in streams affected by mining was observed with species richness (median of 6 in sites downstream of valley fills and 12 in unmined streams, Stauffer and Ferreri, 2002). For example, in Kentucky, Stauffer and Ferreri (2002) observed sites downstream of valley fills had a median richness of 7 fish species, compared to a median richness of 12 fish species in streams without mining in the watershed. In these streams, the number of benthic fish species was also lower in the sites downstream of valley fills (median = 1 benthic species) than in the streams without mining in the watershed (median = 6 benthic species).

Fulk et al. (2003) used the Mid-Atlantic Highlands Index of Biotic Integrity (IBI) to analyze fish data from 27 streams in West Virginia. The index is composed of several metrics that are responsive to environmental and chemical stress, e.g., native intolerant taxa, native Cyprinidae taxa, native benthic invertivores, percent Cottidae, percent gravel spawners, and percent piscivore/invertivores were expected to decrease with increasing stress, and percent macro-omnivore, percent tolerant fish and percent exotic fish were expected to increase with increasing stress. In their study, Fulk et al. (2003) classified streams (e.g., no mining in the watershed, sites downstream of valley fills, mountaintop mining in the watershed, sites downstream of valley fills and with residential development in the watershed) and compared fish metrics and the composite IBI among stream classes. IBI scores from the sites downstream of valley fills were significantly lower than scores from sites without mining in the watershed by an average of 10 points, indicating that fish communities were degraded in sites downstream of valley fills. In their analysis, Fulk et al. (2003) suggest that the reduced index score was caused by fewer minnow species and benthic insectivores in sites downstream of valley fills compared to streams without mining in the watershed. Index scores were also lower at sites with mining in the watershed compared to scores from streams without mining in the watershed. Watershed size was also an important factor in this analysis. Sites with mining and valley fills in small watersheds (<10 km^2) showed more degradation than sites with mining and valley fills in large watersheds (>10 km^2) (Stauffer and Ferreri, 2002; Fulk et al., 2003).[12]

6.2. EFFECTS ON ECOLOGICAL FUNCTION

No studies were found that assessed the impact of MTM-VF on ecological function, e.g., biogeochemical cycling, in downstream habitats. Additional research is needed to better understand the effects of MTM-VF on ecological function in downstream sites.

6.3. BIOLOGICAL CONDITION

The biological effects downstream of MTM-VF are consistent with generic narrative descriptions of moderately to severely degraded condition (Davies and Jackson, 2006). The biological condition gradient (BCG) is a framework that identifies 10 attributes of stream ecosystems indicative of biological status ranging from pristine, natural condition (Tier 1) to severely degraded condition (Tier 6) (Davies and Jackson, 2006) (see Figure 10). Evidence was available to evaluate 3 of the 10 ecological attributes described in the BCG. Sensitive taxa, specifically insect Order Ephemeroptera, are markedly diminished downstream of MTM-VF (Tier 5). The spatial and temporal extent of detrimental effects is between the reach- and catchment-scale (Tiers 4 to 6). Finally, the burial of the headwaters and the construction of channels correspond with a loss of ecosystem connectance between 'some' and 'complete' (Tiers 4 to 6). The attributes identified in the BCG highlight many data gaps—including documenting the extent of regionally endemic taxa, reporting the relative tolerance of taxa to the stressors

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[12]Because larger watersheds typically have greater fish diversity than smaller watersheds, both studies adjusted their analyses to account for the potential effect of watershed size.

specific to the region, identifying the presence of nonnative organisms, reporting the condition of organisms and measuring ecosystem functions in both reference and MTM-VF streams. The BCG provides a general framework and is intended to be locally calibrated by state and regional scientists and resource managers. Local calibration would provide a useful framework for describing the effects of MTM-VF and restoration efforts on stream condition.

6.4. RELATIONSHIP BETWEEN BIOLOGICAL METRICS AND ENVIRONMENTAL FACTORS

Five environmental variables associated with mining and valley fills are commonly considered to potentially affect the ecological condition of downstream habitats: ion concentration, heavy metal concentration, organic enrichment, changes to instream habitat and changes to upstream land use/land cover. This section describes associations between these variables and biological characteristics.

6.4.1. Ion Concentration

All studies report elevated ion concentration in MTM-VF (see Table 14), and many show strong negative correlative relationships between biological metrics and specific conductance and/or the concentrations of individual ions (Howard et al., 2001; Stauffer and Ferreri, 2002; Fulk et al., 2003; Hartman et al., 2005; Merricks et al., 2007; Pond et al., 2008). Though not focused on mountaintop mining effects, elevated conductivity (i.e., 500–8,000 µS/cm, has been shown to have a significant negative correlation with the number of pollution sensitive taxa in benthic macroinvertebrate communities [Soucek et al., 2000; Kennedy et al., 2003]). Ephemeroptera richness was negatively correlated with specific conductivity (Hartman et al., 2005). Though Merricks et al. (2007) did not assess conductivity-macroinvertebrate relationships among sites, they note that only sites with the highest levels of conductivity, ranging between 2,657 to 3,050 µS/cm, lacked Ephemeroptera. Pond et al. (2008) performed the most complete analysis of ions and observed strong negative relationships between specific conductance and their biological assessment measures, GLIMPSS ($r = -0.90$) and WV SCI ($r = -0.80$), total generic richness ($r = -0.74$), EPT generic richness ($r = -0.88$), number of Ephemeroptera genera ($r = -0.90$) and the number of Plecoptera genera ($r = -0.75$). Pond et al. (2008) further demonstrated a decline in number of Ephemeroptera taxa and community proportion when conductivity levels exceeded around 500 µS/cm. HCO_3^-, Ca, hardness, Se SO_4^{2-}, Mg), K^+, and Na^+ were also found to have strong negative correlations with biological metrics (Stauffer and Ferreri, 2002; Pond et al., 2008). While these studies do not provide enough detail to elucidate the mechanistic relationship of biological degradation to ion concentration, the pattern clearly suggests a strong association between the two.

6.4.2. Specific Metals

Though contributing to overall ion concentration, the concentrations of individual metals were negatively correlated with many of the biological metrics in streams. Hartman et al. (2005) found strong negative relationships (Pearson's correlation coefficients ranged from −0.70 to −0.98) between macroinvertebrate metrics and metals. For example, Ephemeroptera richness was negatively related to Cu, Fe, Mn, and Ni; EPT taxa richness was negatively related to Mn and Ni; Plecoptera richness was negatively related to Cd. That study as well as Merricks et al. (2007) reported that metal concentrations were higher in mining streams with biological degradation than in reference streams. These results suggest that elevated metal concentrations associated with mine-impacted streams may contribute to differences in stream biota.

6.4.3. Organic and Nutrient Enrichment

Two studies suggest a possible association between differences in biota and organic enrichment in streams affected by MTM-VF. Merricks et al. (2007) evaluated changes in the composition of functional feeding groups below settling ponds to assess potential organic enrichment. Stations closest to the ponds had significantly higher collector-filterer populations compared to stations further downstream. Merricks et al. (2007) also noted that the HBI was elevated at all fill-influenced sites compared to a reference site. The HBI was developed to respond to a nutrient and organic pollution gradient, but it is also responsive to other stressor gradients, including increased fine sediments and specific conductivity (Paybins et al., 2000). Pond et al. (2008) found moderate to strong correlations between NO_3 and biological metrics (Spearman correlation coefficients ranged between 0.39 and 0.90), but total phosphorus levels were below detection levels at all sites.

6.4.4. Instream Habitat

There was little evidence of an association between changes in macroinvertebrate community metrics and characteristics of instream habitat at sites downstream of MTM-VF. In general, characteristics of macroinvertebrate community composition, such as percent clingers, are predictably affected by stream habitat characteristics (e.g., Pollard and Yuan, 2009). In the case of MTM-VF, most field studies found no systematic relationship between macroinvertebrate metrics and habitat assessment measures (Howard et al., 2001; Hartman et al., 2005; Merricks et al., 2007), which may suggest that habitat characteristics were not all that different between reference and mined stream sites. Similarly, Hartman et al. (2005) did not observe a relationship between macroinvertebrate metrics and fine sediment or turbidity. Individual physical habitat variables and total rapid bioassessment procedure habitat scores were correlated with macroinvertebrate indices in the study by Pond et al. (2008). Because the multimetric indices used by Pond et al. (2008) are sensitive to many factors, the relationship between habitat and macroinvertebrate metrics would be expected. However, they did not observe excessive sedimentation in sites downstream of valley fills.

6.4.5. Disturbance and Loss of Upland Habitat

In addition to the effects described above, changes in upland and headwater areas of the watershed may alter macroinvertebrate composition in downstream habitats. Headwater streams are critical to downstream ecological condition and their alteration, as in mountaintop mining and valley fill activities, may impact the integrity and the sustainability of downstream habitats. Headwater streams provide downstream habitats with water, nutrients, food and woody debris (Gomi et al., 2002; Wipfli et al., 2007). Moreover, the physical structure of headwater streams in the landscape may affect populations and communities of stream organisms by influencing the movement of sediment, chemicals and individuals to downstream reaches within the network (Lowe et al., 2006).

The loss of trophic subsidies from headwater streams may lead to lower secondary productivity in downstream habitats. Food resources, specifically organic matter and macroinvertebrate prey, are transported from headwater streams to downstream habitats. Organic material enters headwater streams through litter fall from riparian vegetation, surface runoff of particulate and dissolved material and subsurface movement (Cummins et al., 1989; Wallace et al., 1999). Once introduced, organic material can be retained in the headwater stream, transformed through feeding of organisms in the headwater stream, or transported downstream (Webster et al., 1999; Wipfli et al., 2007). For example, in a study of 52 streams in Alaska, Wipfli and Gregovich (2002) found that headwater streams were a source of macroinvertebrates and detritus to downstream habitats.

The loss of headwater stream habitat might cause less viable populations of taxa in downstream reaches. Headwater streams sometimes serve as refugia and source areas for downstream biological diversity (Meyer et al., 2007). These areas may facilitate a 'rescue effect' where there is the potential for recolonization from undisturbed sites, and the presence of this source of colonists can be a strong determinant of population resilience (Brown and Kodricbrown, 1977).

Finally, watershed characteristics and activities greatly affect the structure and the function of streams. Houser et al. (2005) showed that intact riparian zones were not sufficient to protect streams from the effect of upland disturbance. They examined the effects of upland soil and vegetation disturbance on ecosystem respiration and found lower ecosystem respiration rates in streams with higher levels of upland disturbance. This is relevant because mountaintop removal represents a significant disturbance to the vegetation and soil characteristics in upland areas. As a result, even when downstream reaches and associated riparian areas of a stream appear intact, as in the case of MTM-VF, they may incur significant impacts from mountaintop removal occurring upstream.

6.5. CUMULATIVE EFFECTS

There is little evidence in the peer-reviewed literature of cumulative impacts of mining on downstream ecology. Fulk et al. (2003) found no evidence of additive effects of multiple mines on the fish IBI. In another MTM-VF study, Pond et al. (2008) reported no evidence of a significant relationship between the number of upstream valley fills and macroinvertebrate indices.

Table 13. Summary of research examining the relationship between mountaintop mining and ecological characteristics in downstream habitats

Reference	Experimental design	Ecological response	Observed effect[a]
Fulk et al., 2003	Fish and benthic macroinvertebrate survey comparing MTM-VF streams (*n* = 21) to regional reference streams (*n* = 5)	Fish IBI	Lower
		Invertebrate IBI	Lower
Hartman et al., 2005	Benthic macroinvertebrate survey comparing filled streams (*n* = 4) to reference streams without valley fill (*n* = 4)	Coleoptera density	Lower
		Diptera density	No difference
		Ephemeroptera density	Lower
		Odonata density	Lower
		Plecoptera density	No difference
		Trichoptera density	No difference
		Total density	No difference
		EPT density	No difference
		Chironomidae density	No difference
		Noninsect density	Lower
		Collector density	No difference
		Scraper density	Lower
		Shredder density	Lower
Howard et al., 2001[b]	Benthic macroinvertebrate survey comparing streams in mined watersheds (*n* = 8) to streams in watersheds without mining activity (*n* = 4)	Taxa richness	Lower
		EPT index	Lower
		Biotic index	Higher
		% clinger	Lower
		% Ephemeroptera	Lower
		% chironomids + oligochaetes	Higher
		KY MBI	Lower
Merricks et al., 2007	Benthic macroinvertebrate survey comparing filled streams (*n* = 4) to a reference stream without valley fill (*n* = 1)	Total richness	No difference
		EPT richness	No difference
		Ephemeroptera richness	Lower
		Plecoptera richness	No difference
		Trichoptera richness	No difference
		Hilsenhoff Biotic Index	Higher
		% Chironomidae	No difference
		% EPT	Lower
		% Ephemeroptera	Lower
		% Plecoptera	Lower
		% Trichoptera	Higher
		% collector-filterer	Higher
		% shredder	Lower

Table 13. Summary of research examining the relationship between mountaintop mining and ecological characteristics in downstream habitats (continued)

Reference	Experimental design	Ecological response	Observed effect[a]
Pond et al., 2008	Benthic macroinvertebrate survey comparing MTM-VF streams (n = 27) to unmined reference streams (n = 10)	Total richness	Lower
		EPT richness	Lower
		Ephemeroptera richness	Lower
		Plecoptera richness	Lower
		WV genus biotic index	Lower
		WV family biotic index	Lower
		Shannon H'	Lower
		% Orthocladiinae	Lower
		% Chironomidae	Lower
		% Ephemeroptera	Lower
		% Plecoptera	No difference
		% EPT	Lower
Stauffer and Ferreri, 2002	Fish communities were compared in filled streams (n = 9) to stream without mining activity (n = 9)	Fish species richness	Lower
		Benthic fish richness	Lower

[a]Comparing the mean values from the reference and downstream and/or mined sites, where lower indicates that the mined/valley fill site has a significantly lower metric values than the reference site (significance as determined by statistical analyses in original study).
[b]The original study did not present statistical analyses on these comparisons.

Table 14. Average ion concentration (reported as specific conductance) in MTM-VF and reference streams reported in conjunction with biological data. Range values are included when reported by the source literature. Standard error values were not reported in the source literature.

Source	Units of measure	Filled		Reference	
		n	Mean (range)	*n*	Mean (range)
Hartman et al., 2005[a]	muhm/s [*sic*]	4	1051	4	150
Howard et al., 2000	µmhos/cm	8	994 (420–1,690)	4	47 (30–66)
Merricks et al., 2007[a,b]	µS/cm	3	1,653 (991–2,720)	1	247
Pond et al., 2008	µS/cm	27	1,023 (159–2,540)	10	62 (34–133)
Stauffer and Ferreri, 2002[c]	µmhos/cm	8	1,716 (513–2,330)	9	164 (125–210)

[a]Averages calculated from reported values.
[b]Values taken from Site 1, which is the first site below valley fill and pond.
[c]Values reported were limited to the Mud River watershed.

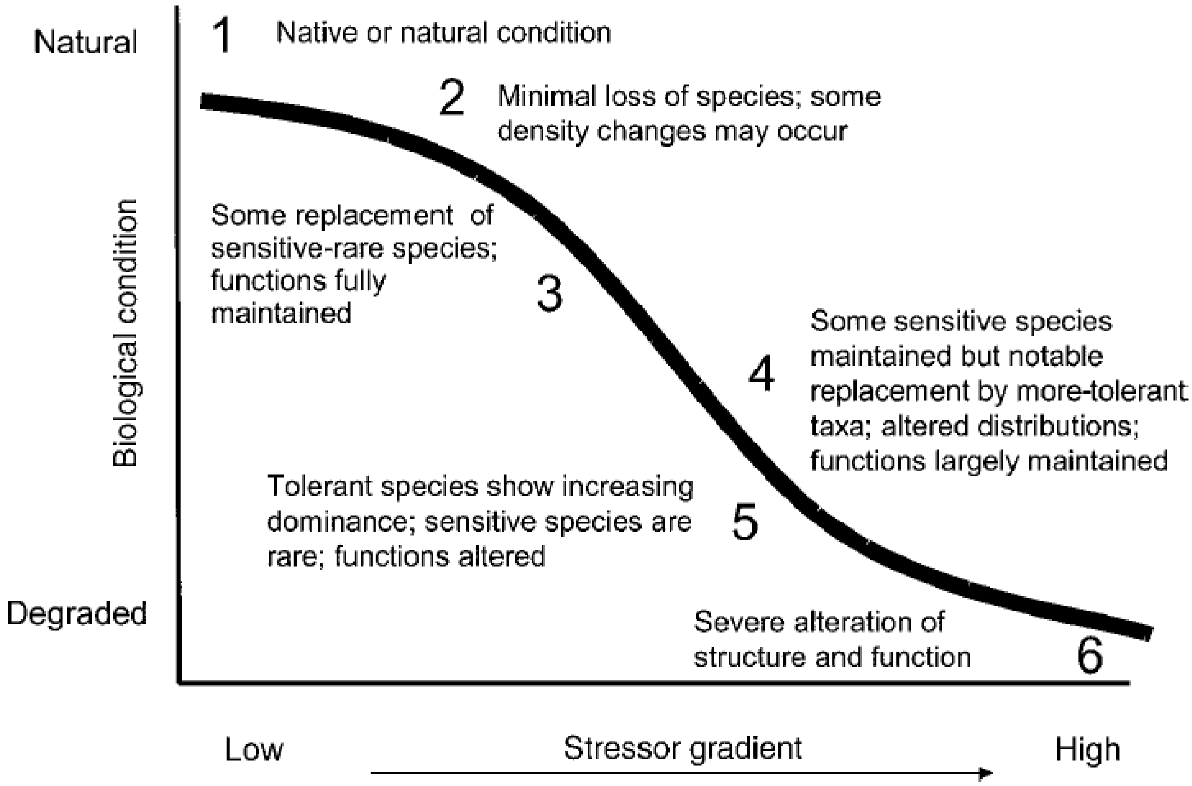

Figure 10. Conceptual model depicting stages of change in biological conditions in response to an increasing stressor gradient.

Source: Davies and Jackson (2006) permission pending.

7. RECLAMATION, MITIGATION AND RECOVERY

Since 1975, surface mining and reclamation have produced significant changes in the topography, hydrology, vegetation, groundwater and wildlife of the central Appalachian Plateau (Loveland et al., 2003; U.S. EPA, 2003, 2005) and have been the dominant driver of land cover/land use change in the Central Appalachian Coalfields (Townsend et al., 2009). Hooke (1999) identified MTM-VF in eastern coalfields as the greatest contributor to earth-moving activity in the United States (see Figure 11).

Under the SMCRA (SMCRA, 30 U.S.C. § 1231, *et seq*.), reclamation is not considered complete until water quality leaving the site complies with CWA standards without additional treatment. Among other things, SMCRA and OSM regulations require that mine operators:

- minimize disturbances and adverse impacts on fish, wildlife, habitat and hydrologic balance,
- recover the approximate original contour (AOC) and vegetation in mined areas and
- restore or approximate the original stream channels and riparian vegetation in permanent constructed stream diversions (30 U.S.C. 1260 and 1265; 30 C.F.R. 816.43, cited in the PEIS, Chapter II [U.S. EPA, 2003, 2005]).

In practice, the ecological conditions at reclaimed/mitigated mined sites vary with the topography and geology of the location, the extent and scale of mining, excavation and reclamation, the techniques used, state and federal regulations in place, goals for the postmining land use, surface property ownership, time elapsed since reclamation and differences in local reclamation or mitigation standards and requirements.

7.1. OVERVIEW OF RECLAMATION AND MITIGATION PRACTICES

7.1.1. Reclamation

Under the SMCRA, reclamation is required to restore lands disturbed by mining to their premining conditions and land uses. Pre-SMCRA (pre-1977) mining practices left large areas of unstable land and eroding hill slopes that impaired streams and created human health risks from mudslides and pollution. That history, plus new concerns about the much larger volumes of blasted rock and debris being produced by then-new mountaintop removal technology, led to early SMCRA enforcement priority on stability and flood control. Reclamation techniques developed prior to 2000 focused on regrading, soil compaction, fast-growing herbaceous vegetation and stabilization, rather than reforestation or stream restoration, for protection of water quality.

Since 2000, reclamation techniques have been developed that seek to restore at least some of the productivity and ecological function of native forests, based in part on research and extension efforts of the Appalachian Regional Reforestation Initiative (ARRI), a coalition of groups formed to promote reforestation of eastern coal mine sites (http://arri.osmre.gov).

7.1.2. Reclamation Bonds

Activation of a mining permit requires posting of reclamation bonds to insure that the coal operator will implement the approved reclamation plan, or provide funds for the government to complete this work should a coal operator forfeit its responsibilities: (SMCRA, 30 U.S.C. § 1259).

Reclamation bonds are released upon inspection in three phases:

- Phase 1: released after backfilling, placement of homogenized or crush-rock topsoils and contour regrading have been completed on a given area.
- Phase 2: released upon completion of revegetation activities.
- Phase 3 (final): released after the mine site has been inspected and accepted as being satisfactorily reclaimed to the approved postmining land use (i.e., meets all performance standards and the approved permit plan) (U.S. EPA, 2003, 2005).

Historically, release of bonds at a given site typically occurred within 5 years after completion of reclamation and was based on percentage of land covered by fast-growing grasses or legumes (Holl and Cairns, 1994; U.S. EPA, 2003, 2005). Reclamation to forested land uses, which are preferred over pasture or hay land uses, may take longer but are more consistent with the SMCRA and CWA (U.S. EPA, 2003, 2005; Skousen et al., 2006).

7.1.3. Mitigation

USACE requirements on Section 404 CWA permits “strive for no net loss of aquatic functions.” This requirement to avoid, minimize, or compensate for unavoidable impacts to waters of the United States has become an important economic factor in mining decisions (U.S. EPA, 2003, 2005).

The USACE uses CWA Section 404(b)(1) to evaluate proposals to convert waters of the United States to dry land (U.S. EPA, 2003, 2005). The preferred alternative is to avoid placing fill in streams; if avoidance is not possible, fills must be minimized “to the extent practicable.”

In either case, the proposal must not result in significant degradation to waters of the United States. Unavoidable impacts to waters are allowed if they can be offset by mitigation that compensates for the aquatic ecosystem functions lost in conversion (U.S. EPA, 2003, 2005).

Mitigation plans are developed prior to the start of mining and involve the use of stream assessment methods to evaluate stream quality before and after impact. Compensatory mitigation requirements are intended to minimize individual and cumulative mining impacts by incentivizing applicants to avoid the delay and increased need for financial assurances or bonding to guarantee that mitigation will be completed (U.S. EPA, 2003, 2005). Fills resulting in "minimal impact, individually and cumulatively, including compensatory mitigation" as determined by local regulatory agencies and the USACE, can be authorized by a Nationwide Permit to expedite the review process. Fills causing more than minimal impact undergo a more detailed individual review. Further, the cost of compensatory mitigation is higher, and permits are at greater risk of being delayed or denied when "high value" aquatic resources are at stake.

Mitigation can be on-site or off-site of the permit area. On-site mitigation is preferred, but off-site mitigation may be the only option for lost mountaintop ecosystems and is common in MTM-VF permits. Compensatory mitigation plans include a variety of actions, including

- Riparian restoration (e.g., revegetation, wetland creation, floodplain connectivity).
- Creating channels to replace channels that have been filled using natural stream design techniques.
- Enhancing or improving existing stream channels (e.g., riffles/pools, dredging, sinuosity, bank stabilization).
- Sediment and pollution control (e.g., reclamation of abandoned mine lands and remediation of other adverse environmental conditions within the watershed, anoxic limestone drains, drums, flumes and other passive treatment systems).
- Reforestation of areas adjacent to mining sites.
- Improving fisheries habitat (e.g., shading, increasing habitat heterogeneity, aeration through riffles or other natural means).
- Removing stream encroachments (e.g., roads, crossings, ponds, or other fills).
- Removing large woody debris.

We discuss only the practices and effectiveness of on-site mitigation for MTM-VF disturbances. A discussion of off-site mitigation, including wetland creation, may be found in Chapter III and Appendix D of the PEIS (U.S. EPA, 2003, 2005).

7.2. EVIDENCE OF RECOVERY: VEGETATIVE COVER

Vegetative cover at most reclaimed mine sites consists of rapidly growing grasses and legumes, which for decades have served as the low cost, low-risk option for reclamation bond release (Loveland et al., 2003). Even when permitted reclamation plans call for the planting of trees, in response to SMCRA, reclamation activities typically result in excessive compaction of the rooting medium, which reduces erosion but severely reduces tree growth (Holl, 2002; U.S. EPA, 2003, 2005; Skousen et al., 2006).

Non-native grasses and legumes are used for reclamation at most sites (U.S. EPA, 2003, 2005, Table 3.J-1). Over time, native plant species do move onto reclaimed mine sites. In a study of 18 reclaimed mine sites in Virginia, Holl and Cairns (1994) found that the total number of plant species was increasing through time. However, succession rates were slower and plant biodiversity greatly reduced relative to forests at nearby unmined sites. Of the 80 plant species occurring in unmined forests, after 2−30 years, only 38 (47%) were found at one or more mined/reclaimed sites, possibly due to the loss of seed banks in stripped topsoils.

Soil compaction and the initial planting of nonnative grasses interfere with the long-term ecological recovery of mined sites. Holl (2002) resurveyed 15 mined sites and 5 logged sites from the earlier study and found that the variety of plant species at the mined sites was more similar to the unmined sites than during the first study. However, native plant biodiversity was still substantially lower than in natural forests. Trees and woody shrub diversity was reduced by 45%, and herbaceous plant diversity was reduced by 56%, compared with forests at unmined sites.

Pasture, grasses and hay lands planted to meet the legal requirements of reclamation may be viable while maintained but may collapse when agronomic practices are neglected after release of bond (Barnes, 2003). Reclaimed areas seeded with grasses and legumes plants will eventually be recolonized by native plants as the nonnative grasses and legumes gradually decline, but the time frame for recovery is not clear. Establishment of mid- to late-successional trees may take decades (Skousen et al., 2009).

The combined effect of forest fragmentation and herbaceous reclamation planting may undermine the ecological condition of unmined forest sections adjacent to mining sites as well. Handel (2001) surveyed 55 mountaintop mining sites in southern West Virginia that were reclaimed with herbaceous vegetation 6−24 years earlier and determined that both abundance and species richness of trees and shrubs were extremely low on mine sites compared to surrounding forests. Further, nonwoody plant species richness was reduced on plots adjacent to mountaintop mining sites compared with interior forests. This edge effect extended from the reclamation area 50 m into the adjacent forest.

As a result of these studies and reports, reforestation of mined land is emerging as the "best practices" postmining land use option, especially in mined areas that previously supported high-quality natural forests (U.S. EPA, 2003, 2005). Although forestry and soil research is being done at government, mining and university extension and experiment stations, little has been published on the subject in the peer-reviewed literature. One exception is a study comparing different conditions for the reforestation of reclaimed surface mined lands in the Appalachian coalfields by Casselman et al. (2006), who found that tree survival and growth rate were affected by differences in the composition of minesoils, the tree species planted and silvicultural methods selected.

7.3. EVIDENCE OF RECOVERY: MINESOILS

Arguably the most limiting factors in the successful reclamation of vegetative cover at a site are the density and composition of minesoils. Minesoils are the mixtures of soils, debris and fractured rock overburden that are spread on reclaimed surfaces to support plant growth (U.S. EPA, 2003, 2005). Natural soils develop over thousands of years as a result of physical and chemical weathering and biological activity on parent rock substrates (Sencindiver and Ammons, 2000). Minesoils are synthetic topsoils created by mixing natural topsoils and crushed rock with soil amendments (fertilizers, biosolids, manure, coal combustion by-products, or mulches) to enhance organic matter. Challenges to the recovery of native vegetative include loss of natural soil structure and the loss of native seed pools and microbial communities in the mining process.

Compaction of minesoils with the use of heavy equipment during valley fill and reclamation is identified as one of the chief factors reducing the value of reclaimed forest lands. Compacted soils limit root growth and greatly reduce the growth and survival of trees (e.g., Skousen et al., 2009). Soil research suggests that compaction below the plow layer may persist a century or more (U.S. EPA, 2003, 2005). Because so much mine area has been reclaimed with compacted soils and heavy herbaceous ground cover, converting such areas to a commercially valuable forest requires expensive tilling techniques such as bulldozer "ripping" (for loosening compacted minesoils), application of herbicides (to reduce the ground cover competition) and use of weathered "brown" sandstone in substitute topsoils (Skousen et al., 2009).

Minesoils reflect the geochemical environment of the blasted rock used to make them and the amendments added to them. Almost all synthetic or substitute soils at MTM-VF sites are characterized by low soil organic carbon (SOC) content and low fertility. Most minesoils develop weak to moderate structure in their surface horizons in as little as 2 years (Sencindiver and Ammons, 2000). Rapid replenishment of soil carbon levels was documented by Simmons et al. (2008), who estimate that normal levels of SOC could be achieved as quickly as 20–50 years after reclamation, although estimates from other studies are considerably longer (discussed below).

Low SOC levels are correlated with high carbon sequestration rates in minesoils. Ussiri and Lal (2005) reported that SOC sequestration rates in the top 15 cm of minesoil are high in the first 20 to 30 years after reclamation, and that higher rates of SOC sequestration were observed in soils (excluding aboveground and leaf-litter storage, where most carbon storage in forests exists) at pasture and grassland mine sites than forested sites. They propose that minesoils have potential for resequestering some of the carbon (C) lost through the soil disturbance associated with deforestation.

Hardwood forests sequester large amounts of carbon for long (>100-year) periods of time (Ussiri and Lal, 2005). Unlike pasture and croplands, forests store large amounts of carbon in litter layers and aboveground biomass, as well as soils. Sperow (2006) found that potential C storage rates are highest in mined lands that are reclaimed directly to forests, as opposed to pasture, crops, or natural forest succession. This study estimated potential C sequestration for 11 km^2 of mine permit land in seven states, including the Central Appalachian Coalfields, which showed mining activity since 1992. Reclaiming all land directly to forests provided an average total sequestration potential of 29.1 Tg C over a 20-year period, with the largest sequestration potential in West Virginia (10.1 Tg C). West Virginia has high sequestration potential because of the dense forest cover and large area of mining permit land in that state. By comparison, average potentials for pasture and croplands over the 11-km^2 area were 16.3 and 10.9 Tg C over 20 years. Although reclaimed forests do not mitigate for the CO_2 emissions of deforestation, especially in mature hardwood forests, differences in reclamation practices can significantly affect projected greenhouse gas emissions from mining regions (Sperow, 2006).

Amichev et al. (2008) measured carbon sequestration in mined and unmined forests in the Midwestern and Appalachian coalfields. Unmined hardwood forests sequestered 62% more carbon than mined forests of any type. Most of this difference (~60%) was in the soil carbon content. Although carbon-poor minesoils have high potential for carbon sequestration, the realization of this potential may have a long lag time (>60 years, with estimates ranging up to 760 years, reviewed in Amichev et al., 2008).

7.4. EVIDENCE OF RECOVERY: WATER QUALITY, STREAM ECOLOGICAL FUNCTION, AQUATIC BIOTA

Sections 3, 4, 5, and 6 have reviewed the impacts of MTM-VF on water quality and aquatic ecosystems. In this section, we provide an overview and discuss the effectiveness of on-site reclamation activities.

7.4.1. Long-Term Effects on Downstream Water Quality

The field studies of downstream conditions reviewed in Section 4, *Impacts on water quality,* were conducted 3 to 24 years after permitting, and field studies reviewed in Section 6, *Impacts on aquatic ecosystems* were done 3 to 15 years after reclamation. The results indicate that reclamation and mitigation, while providing stabilization, flood control and partial recovery of terrestrial soils and vegetation, do not eliminate the deleterious effects of dissolved chemical ions associated with effluent from MTM-VF.

7.4.2. Created Headwater Streams

Thousands of kilometers of headwater streams have been buried or removed by mountaintop removal, and there is no substantive evidence in the literature or PEIS that onsite mitigation by constructed channels or wetlands has replaced or will replace the lost functions and biodiversity. Appalachian headwater streams are integrally associated with the geologic strata, aquifers, subsurface flows, soil layers, surface topography and vegetation of the parent mountain (see Figure 3). When a mountain is leveled, the geologic structures, groundwater exchanges, hillslopes and subsurface flows that supported its small streams are permanently dismantled. Stream restoration techniques were developed to restore one or more features of an existing stream that has the basic structure of terrestrial and subsurface ecosystems intact, not to create streams starting from scratch (Palmer, 2009).

The combination of ditches, groin ditches and sediment retention ponds are designed to convey runoff for large (e.g., 100-year) storm events. No credible evidence was found to support claims that the constructed channels and diverted surface flows on valley fills restore the diversity and functions of natural headwaters lost to mining. Evidence that the constructed channels and ponds are supporting any aquatic communities is scarce (Green et al., 2000), but the small data sets available show that assemblages colonizing reclaimed surface waters differ from natural headwater assemblages in predictable ways: loss of headwater-specific taxa; increased ion- and sediment-tolerant taxa; and presence of taxa adapted to ponds or turbid, slow-moving water (Kirk, 1999). Local habitat and biodiversity of stream ecosystems are strongly influenced by the land use and land forms within the surrounding valley at multiple scales (Allan, 2004), so constructed channels from which all terrestrial and subsurface ecosystems have been removed are likely to differ from the natural systems they mimic.

With some exceptions, restoring ecosystems at mined sites to their premining condition is not a realistic goal when mining methods destroy the underlying structure of those ecosystems. To recover deep coal layers, mountaintop mining operations remove up to 1000 vertical feet of unweathered rock by blasting. Much of this crushed rock is moved to the valleys to construct the fills (see Figure 3). When mountaintop and coal removal are complete, the new landscape has dramatically different elevations, contours and geology, all of which profoundly influence ecological conditions at a site. Created channels on regraded contours provide channelized flows, and some even provide limited habitat for aquatic invertebrates. However, surface and subsurface conditions are so different that even with stream restoration techniques far better than those available today, flow channels constructed at these sites cannot reasonably be expected to restore or mitigate for the premining condition of aquatic life or water quality.

Flood control is an important function in the modified postmining landscape, as increased stormflow volumes have been observed (U.S. EPA, 2003, 2005). Ferrari et al. (2009) modeled the flood response in the 187-km^2 mined watershed of Georges Creek in Western Maryland and found parallels to what would be expected for urban areas with large areas of impervious surface. Infiltration rates in reclaimed sites are typically 1 to 2 orders of magnitude smaller than for undisturbed forest (Negley and Eshleman, 2006). Ferrari states that, "As a consequence, the act of mine reclamation should not be interpreted as meaning the land is returned to a state that is the hydrological equivalent of the premining landscape."

In a 2008 Federal Register notice, the USACE clarified the need to identify streams as "difficult-to-replace" aquatic resources and acknowledged the need to avoid and minimize impacts to them:

> We recognize that the scientific literature regarding the issue of stream establishment and re-establishment is limited and that some past projects have had limited success (Bernhardt et al., 2007). Accordingly, we have added a new paragraph at 33 C.F.R. 332.3(e)(3) [40 C.F.R. 230.93(e)(3)] that specifically notes that there are some aquatic resources types that are difficult to replace and streams are included among these. It emphasizes the need to avoid and minimize impacts to these 'difficult-to-replace' resources and requires that any compensation be provided by in-kind preservation, rehabilitation, or enhancement to the extent practicable. *This language is intended to discourage stream establishment and re-establishment projects while still requiring compensation for unavoidable stream impacts in the form of stream corridor restoration (via rehabilitation), enhancement and preservation projects, where practicable* [italics added for emphasis].

Differences in the actual size of Appalachian headwaters (see Table 15) and the impact threshold drainage area set for mitigation indicate that difficult-to-replace resources may be overlooked in the planning stages of mine development. The mitigation threshold is determined by local authorities and typically set to a drainage area ranging from 0.8–1.9 km^2 (200−480 acres). Drainage areas for headwaters in the PEIS study range from 0.03–0.61 km^2 (6−150 acres) (see Table 15).

Table 15. Drainage areas for headwaters in the PEIS study area (KY, TN, WV, VA)

Headwater drainage type	Median drainage area km^2 [acres]	Range of drainage areas km^2 [acres]
Ephemeral-intermittent	0.06 [14.5]	0.03 to 0.18 [6.3 to 45.3]
Intermittent-perennial	0.17 [40.8]	0.04 to 0.61 [10.4 to 150.1]

Source: Paybins (2003).

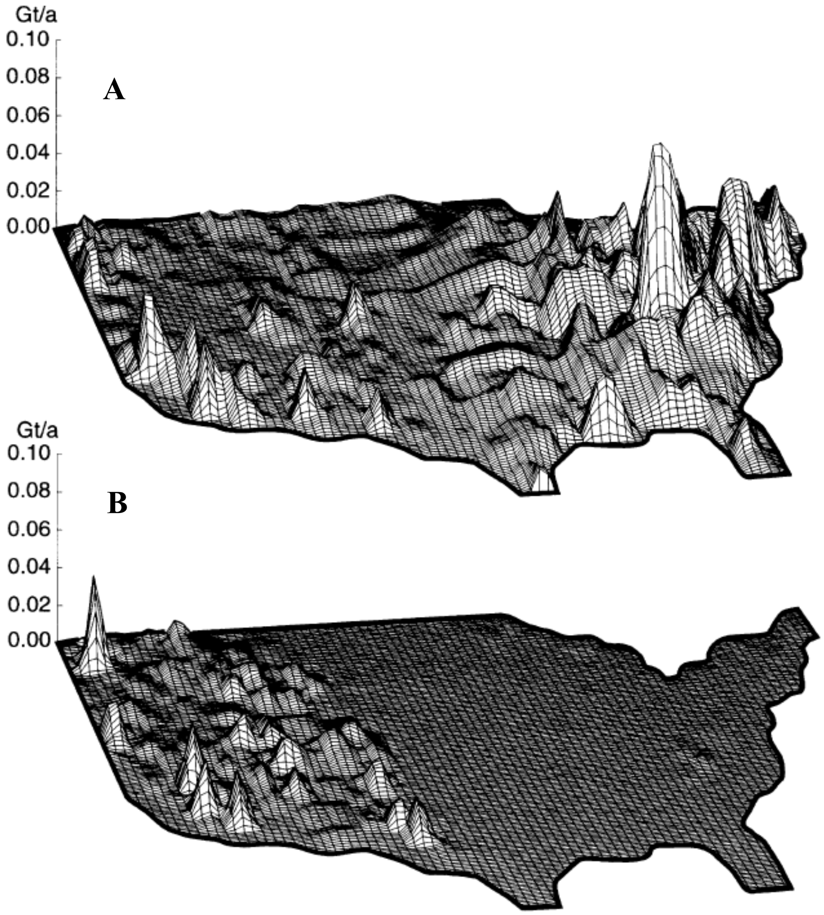

Figure 11. Earth movement by humans and streams. Maps of the United States showing, by variations in peak height, the rates at which earth is moved in gigatonnes per annum in a grid cell measuring 1° (latitude and longitude) on a side, by (A) humans and (B) rivers.
Source: Hooke (1999), used with permission from the publisher.

8. SUMMARY, INFORMATION GAPS AND RESEARCH OPPORTUNITIES

8.1. A CONCEPTUAL MODEL OF THE IMPACTS OF MTM-VF

Mountaintop mining is a type of surface mining that is currently used, particularly in the Central Appalachian Coalfields, to mine relatively low-sulfur coal. The subject of this review, mountaintop removal mining, differs from other types of coal surface mining (e.g., contour strip mining) in its scale. Whereas contour mining removes overburden along a bench cut into the side of a mountain to expose a coal seam and then removes the coal, mountaintop removal operations use explosives to remove all or portions of the entire mountain or ridge top, as much as 300 vertical meters (1,000 vertical feet) of overburden, to expose and mine one or more coal seams.

Aquatic ecosystems downstream of MTM-VF differ in significant ways from streams that receive little human influence. Observations of which organisms are lost and how the streams they live in are altered can improve our understanding of how MTM-VF result in these impacts. Our review of the peer-reviewed, published literature and the PEIS is summarized here in a conceptual model that traces the impacts of MTM-VF on aquatic ecosystems (see Figure 12).

Impacts begin with the preparation of the mountaintop or ridge top site. Access roads are built, all trees are cleared and some topsoil is stockpiled for future use in reclamation. Then, explosives are used to blast the top of the mountain or ridge to expose and mine one or more coal seams. The coal that is removed is processed and transported to market; we did not encompass the impacts of these processes in our review. Instead, we follow the fate and impacts of the excess overburden and the mined site that remains.

When the mountaintop is removed, so are the intermittent streams, springs and small perennial streams that comprise the headwaters of rivers. The excess overburden is disposed of in constructed fills in small valleys or hollows adjacent to the mountaintop site. When trees are removed from the valley fill footprint prior to construction of the fill, it also removes habitat for amphibians that move between forest and stream during their life cycle.

When the valley fill is constructed, the headwaters beneath the footprint are buried, and most organisms that lived there are killed. These headwaters support diverse biological communities of aquatic invertebrates, such as insects, and vertebrates, including fish and salamanders, that are often distinct from the species found in further downstream in the stream system. Permanent headwater reaches can be spawning and nursery areas for native brook trout, *Salvelinus fontinalis*. Intermittent and permanent headwater reaches, particularly those too small to support fish, support numerous amphibian species. This particularly includes salamanders, of which nearly 10% of the global diversity is found in streams of the southern Appalachians.

These streams are also habitat for diverse assemblages of Ephemeroptera, Plecoptera, Trichoptera and other aquatic insects, many of which are unique to these headwater reaches. Again, this biodiversity is lost when these headwater streams are buried under valley fills. The hyporheic habitat is also buried, eliminating the interface of groundwater and surface water that harbors the microbial community responsible for much of the nutrient processing and increasing the export of nitrogen downstream.

As multiple streams in a mountain range are buried, the distance between the headwaters that remain becomes greater. This hinders the movement of biota that is required to sustain populations.

Both the water and sediment discharged into downstream ecosystems are altered by MTM-VF. Water runoff is increased when the forest is cleared for the mine and valley fills. The compacted, bare soils, which result from the removal of the overburden and coal, form a large impervious surface that increases surface runoff. Depending on the degree to which they have been compacted, the valley fills can ameliorate the effects of moderate precipitation events on high flows, because they temporarily store water from surface flows and direct precipitation. However, precipitation from more intense storms may produce greater storm flows, because compaction of the fill surface and mined area by heavy equipment may reduce infiltration of precipitation and increases overland runoff.

Surface runoff is diverted into ditches and sedimentation ponds, replacing natural subsurface flow paths. Under most circumstances, the sedimentation ponds are effective at settling out fine sediments. However, organisms that cling to rocks in riffles were observed to decrease downstream of the ponds, a finding associated with increased fine sediments in other ecosystems. The ponds themselves change the predominant source of energy in the downstream systems from tree leaves to algae. Organisms that feed on leaves (shredders) decline; organisms that feed on algae (filterers) increase.

The overburden in backstacks and valley fills has increased surface area available for water contact with rock particles, and the water that emerges has higher concentrations of major ions and some trace metals, including selenium. Native mayflies are consistently among the first to disappear as concentrations of ions and trace metals increase. Most studies have found strong negative correlations between the biotic metrics for fish or macroinvertebrates and specific conductance, total dissolved solids, the concentration of individual ions, like SO_4^{2-}, and other measures of the elevated concentrations of various ions observed in streams below valley fills. These studies have also generally found negative correlations between the various biotic metrics and some measures of metals, but trace metals are generally less elevated in streams below valley fills than in other mining situations and are usually correlated with greater increases in other dissolved ions.

Discharges with high concentrations of ions reduce reproduction and survival as shown in standard toxicity tests using organisms such as the crustacean *Ceriodaphnia dubia*. Concentrations of selenium in some streams have been measured at levels that have been shown to cause fish deformities and reduced fish reproduction in other streams. Fe and Mn deposits have been observed on invertebrates, suggesting that concentrations are elevated under some circumstances. Ni concentrations in sediments downstream of MTM-VF exceed empirical screening values.

The loss of the headwaters has ramifications for the ecosystems downstream. The loss of headwater invertebrate taxa removes a source of food. Fish that specialize in feeding on invertebrates (the benthic invertivores) decrease downstream. Headwaters are active sites of organic matter processing and nutrient uptake. The loss of headwater invertebrates and microbial communities reduces dissolved carbon exports, an important food resource for downstream biota, and increases nutrient loads in downstream waterbodies.

After the coal is removed, the extraction area and valley fill are graded and planted to control sediment runoff. There is evidence that erosion is partially controlled and this mitigates but does not completely eliminate the amount of fine sediments deposited downstream.

8.2. CONCLUSIONS

This section summarizes our major conclusions of potential consequences of MTM-VF in six categories:

(1) Loss of headwater streams and forests
(2) Impacts on water quality
(3) Impacts on aquatic toxicity
(4) Impacts on aquatic ecosystems
(5) Cumulative impacts of multiple mining operations
(6) Effectiveness of mining reclamation and mitigation

We formed our conclusions by reviewing evidence from two sources of information: (1) the peer-reviewed, published literature and (2) the PEIS and its associated appendices (U.S. EPA, 2003, 2005). Only a few peer-reviewed papers were found that studied water quality or stream ecosystems in headwaters or downstream of MTM-VF in the Central Appalachian Coalfields. Our conclusions are based on evidence from these papers and relevant research findings from laboratory studies and observational studies from other locations and mining activities. We also discuss the findings published in the PEIS, which was published as two separate documents; the Draft, published in 2003, and the Final, published in 2005.

8.2.1. Loss of Headwater and Forest Resources

Based on permits approved from 1992 to 2002, more than 1,900 km of headwater streams are scheduled to be permanently lost or buried because of MTM-VF. These streams represent more than 2% of the stream miles in the study area (KY, TN, WV, and VA), and their total length is more than triple the length of the Potomac River. The area of valley fills based on permit approved from 1985 through 2001 was 1774.4 km^2. Additionally, 1,540 km^2 of forests were lost to MTM-VF from 1992−2002. More current statistics were unavailable at the time this report was written. However, by 2012, the area that will have been deforested by MTM-VF in the 4-state study area has been projected to be 5,700 km^2, about 1.4 times the size of the state of Rhode Island.

8.2.2. Impacts on Water Quality

Changes in water quality observed in streams downstream of MTM-VF include alteration of flow and temperature regimes, increased fine sediments and increases in ions, some metals and nitrogen.

Flows in streams below valley fills were generally more constant in both discharge and temperature than in unimpacted streams. Valley fills influence downstream water quality by acting like aquifers that store at least some of the water that enters from groundwater, surface drainage, or direct precipitation. The removal of vegetation, particularly plants that have deep roots, from the mined area and the area covered by the fills increases flow by decreasing evapotranspiration.

Valley fills ameliorated the effects of moderate precipitation events on high flows, likely because they temporarily store water from surface flows and direct precipitation. However, there is some evidence that precipitation from more intense storms results in greater storm flows, because of compaction of the fill surface and mined area by heavy equipment that reduces infiltration of precipitation and increases overland runoff.

Effluent waters below valley fills were often alkaline. Most valley fills contain sufficient carbonate minerals to neutralize the acid produced by pyrite oxidation that has historically caused water quality problems from coal mining. In addition, the sediment retention ponds can be used as treatment basins to neutralize pH. As a result, the metals that are not soluble under higher pH conditions, such as Fe, Cd, Cr, Cu, Pb, Zn, and Al were generally not elevated in effluent waters below valley fills. Under some conditions, such as during higher flows, particulate forms of less soluble metals, such as Fe, may be washed out of the valley fills.

Other ions were consistently observed at greatly elevated concentrations in the discharges from valley fills. SO_4^{2-}, HCO_3^-, Ca^{2+}, and Mg^{2+} are the dominant ions, but others include K^+, Na^+, and Cl^-. These ions all contribute to the elevated levels of total dissolved solids, typically measured as specific conductivity observed in the effluent waters below valley fills. Selenium concentrations are also elevated. Selenium can bioaccumulate through aquatic food webs, and elevated levels have been found in fish tissues of the mining region. More than half of the sites surveyed exceeded the chronic AWQC for selenium. Selenium reaches concentrations that have been associated with effects in fish and birds in studies of mining effluents from other regions.

Despite the construction of sediment retention ponds below valley fills, several studies found increased fine sediments in stream reaches downstream. Concentrations of NO_3^- plus NO_2^- were also slightly elevated downstream.

8.2.3. Toxicity Impacts on Aquatic Organisms

Results of laboratory toxicity tests using the crustacean *Ceriodaphnia dubia* predict that acute lethality will occur at the high end of specific conductivity observed downstream of MTM-VF operations. This expectation was confirmed by reduced survival of *Ceriodaphnia* observed in short-term tests using water from sites with high specific conductivity. Laboratory tests of major ions reported effects on reproduction at concentrations 2−3 times lower than effects on survival. Evidence from other alkaline mine effluents suggests that effects to organisms should be expected at concentrations below those that affected *Ceriodaphnia*. Tests using the mayfly *Isonychia bicolor* and the amphipod *Hyalella azteca* reported effects on survival at concentrations 3−4 times lower than those affecting *Ceriodaphnia*. If effects on reproduction in these organisms are similarly more sensitive than survival, effects from ions would be expected at most sites downstream of MTM-VF.

Results from laboratory studies that varied the mixture of ions indicate that the interplay among ions is complex. The ion mixture reported downstream of MTM-VF sites is dominated by SO_4^{2-}, Ca^+, HCO_3^-, and Mg^+. The low concentrations of Cl^- are likely making effects more severe. The high overall hardness may be a mitigating factor. If the relative proportions of ions are generally consistent downstream of MTM-VF in the Central Appalachian Coalfields, specific conductivity (μS/cm) may be the best surrogate indicator to use to predict when adverse effects would occur.

Se concentrations reported from waters in the study area were high enough to warrant concern. In some streams, they exceeded the chronic AWQC and fall in the range of concentrations that have caused toxic effects on aquatic invertebrates, fish and birds in the field, including in waters receiving valley fill and overburden dump leachates at coal mines in Canada. Although the bioavailability of selenium is difficult to predict, measurements of selenium in fish tissue indicated that the selenium is elevated in a form that is bioavailable.

Other toxicants were also high enough to warrant further investigation. Fe and Mn deposits have been observed on macroinvertebrates. Ni and Zn concentrations in sediments are higher than empirical screening level values.

8.2.4. Impacts on Aquatic Ecosystems

All surveys that used multimetric and aggregate taxonomic indices reported degraded biological conditions in streams downstream of MTM-VF. Both fish and macroinvertebrate communities were affected. Within the communities, changes were observed in organisms grouped by food source: benthic macroinvertebrates that feed on leaf detritus declined, benthic macroinvertebrates that feed on algae increased and specialist fish that eat benthic macroinvertebrates declined. Changes were also observed in organisms grouped by habit; macroinvertebrates that cling to rocks in riffles declined. All studies showed a reduction in mayflies. Trends observed for other taxonomic groups were less striking. Declines of the aggregate indices and mayflies were most strongly correlated with increased concentrations of ions and selenium.

8.2.5. Cumulative Impacts of Multiple Mining Operations

Few studies were found that investigated the cumulative impacts of multiple mining operations. Specific conductivity and SO_4^{2-} levels were elevated in larger streams of the Kanawha basin, downstream of multiple mining operations. Concentrations increased between 1980 and 1988 as more areas were mined. However, based on the results of one analysis, fish or macroinvertebrate multimetric indices did not decline further as the number of upstream mines or valley fills increased.

8.2.6. Effectiveness of Mining Reclamation and Mitigation

Nonnative grasses and legumes were used for reclamation at most of the mines studied. Over time, the total number of native plant species increased but remained less than half the diversity found at nearby unmined sites. Plant species richness in forests adjacent to mining sites also remained lower than interior forests. By current reclamation standards, return of mature forests to mined areas may require millennia due to soil compaction and loss of organic matter.

The results of the water quality studies indicate that reclamation efforts partially controlled the amount of soil erosion and fine sediments transported downstream. However, there is no evidence that reclamation efforts altered or reduced the ions or toxic chemicals downstream of valley fills. Ion concentrations have either remained constant or increased over time. Given that the alterations of the stream ecosystems reported above were observed after sites were reclaimed for 3 to 15 years, the observed effects would be expected to persist.

No published studies were found that investigated the replacement of stream function or biodiversity in stream channels constructed on top of valley fills. However, constructed channels at locations experiencing far less disturbance than mountaintop mining are consistently found to differ significantly from natural systems in biota and functions.

8.3. INFORMATION GAPS AND RESEARCH OPPORTUNITIES

The evidence in our review is consistent enough that we have a high degree of confidence in our conclusions. Still, our review uncovered a number of information gaps that could be filled by research. Filling these can improve our quantitative understanding of how MTM-VF impacts aquatic ecosystems, potentially leading to more effective regulatory and mitigation approaches.

Future assessments should consider the comparative risks of MTM-VF. However, the comparisons should be to real alternatives that might be implemented by real decisions. For example, risks from MTM-VF might be compared with those associated with other coal sources, or risks associated with electricity generated by burning coal from MTM-VF might be compared to those associated with sources of electricity other than coal combustion. Alternatively, if a land use decision is being made, MTM-VF risks might be compared with other uses of mountain tops and headwater streams such as logging or tourism.

8.3.1. Update the MTM-VF Inventory and Surveys of Impact Extent

The most recent data available in the published literature on the extent and potential additional development of MTM-VF mines in the Central Appalachian Coalfields are those compiled for the PEIS (U.S. EPA, 2003, 2005). These data were only for MTM-VF mines developed between 1985 and 2002, when at least some mines had been developed as early as 1967 (U.S. EPA, 1979), and permitting and construction of MTM-VF have continued since then. Therefore, the inventory of filled valleys and of stream miles buried by those valley fills should be updated. Moreover, the inventory should be adjusted to reflect the actual extent of these valley fills versus their permitted extent. Remote sensing and GIS, combined with field sampling, would make this possible (Townsend et al., 2009). The updated inventory of

MTM-VF can be used to design a statistically robust estimate of the extent of impacts within the region, based on a probabilistic sampling design.

8.3.2. Quantify the Contributions of Headwater Streams

It would be desirable to more fully understand the role of the headwater streams buried by valley fills in the retention and cycling of nutrients, such as nitrogen and phosphorus, and the downstream transport of trophic resources, such as allochthonous organic matter (i.e., leaf litter, small particulate organic matter produced from the leaf litter, and dissolved organic carbon), algae and animal prey. This understanding would allow assessors to better understand and model the cumulative effects of burying these headwater streams on stream function (i.e., nutrient transport and cycling, processing and transport of organic matter) and other ecosystem services supplied by these stream systems.

It would be desirable to more fully understand the metapopulation and metacommunity linkages among different headwater streams and between these headwaters and downstream reaches. This information would increase understanding of the effects of burying these headwater streams on regional biodiversity, including the cumulative effects of this practice of burying headwater drainages in this region of the Appalachians.

8.3.3. Improve Understanding of Causal Linkages

Our understanding of the causal linkages between MTM-VF and stream ecosystems could be improved by bringing together additional data. Sources of data include reports that we were unable to include in this report because we could not confirm that they had been peer-reviewed and additional monitoring data that may also be available from various states, particularly West Virginia and Kentucky. Questions that might be answered include

(1) At what concentrations of major ions and trace metals do different taxa disappear?

(2) Which downstream organisms in addition to Ephemeroptera are most affected by valley fills?

(3) How do these effects differ among different insect orders and between insects and noninsect aquatic taxa?

(4) How do the species within these large orders change? Some evidence indicates that headwater species are replaced by more downstream species below valley fills.

(5) Are there observable effects on individuals of sensitive taxa?

When selecting such monitoring data, care should be taken that the sampling was timed so that the common species can be sampled if they are present. This is not a concern for fish, but many macroinvertebrates are present for part of the year as eggs or larval instars that are too small to be sampled by the standard net mesh sizes used to sample benthic macroinvertebrates.

8.3.4. Develop Tests Using Sensitive Taxa

Although the field observations of taxa declines agree qualitatively with the toxicity test results of MTM-VF effluent using standard laboratory organisms, effects to some native organisms appear to be occurring at lower concentrations. Quantitative estimates of the concentrations at which effects occur could be improved by testing effluents using a life-cycle test, especially with vertebrate and invertebrate species found in these headwater streams. For invertebrates, we would recommend an Ephemeroptera species or a physiologically similar aquatic insect. An example of a full life cycle with a species of Ephemeroptera is described by Sweeney et al. (1993) and Conley et al. (2009). Tests using these insects would help verify that the differences in sensitivity between laboratory tests using *Ceriodaphnia dubia* and field observations of Ephemeroptera declines are due to differences in sensitivity to the ions, rather than a confounding factor. For fish and amphibians, it would be desirable to perform reproductive toxicity tests with waters like those found below valley fills using headwater taxa, such as dace, brook trout, or sculpins.

The vertebrate and invertebrate fauna found in headwater streams of the southern Appalachian Mountains are adapted to waters characterized by low hardness, total dissolved solids, ionic strength, conductivity and alkalinity and by neutral to slightly acidic pH. The streams below valley fills are altered such that the waters are characterized by high hardness, total dissolved solids, ionic strength, conductivity and alkalinity and slightly alkaline pH. These waters also have relatively high concentrations of individual ions, such as SO_4^{2-}, HCO_3^{-}, and K^{+}. These multiple changes in the dissolved constituents in these waters are likely to have interactive effects on aquatic fauna and are not duplicated well by any laboratory test data found in the published literature. Moreover, the species and life stages used in the laboratory tests found in the published literature differ from the native fauna of these streams.

Most of the invertebrates that have been used in laboratory toxicity tests of the effects of conductivity, total dissolved solids, or the individual effects of SO_4^{2-} and other dissolved ions are Crustacea. This includes the cladocerans, *Ceriodaphnia dubia* and *Daphnia magna*, and the amphipod, *Hyalella azteca*. These Crustacea have very different evolutionary histories compared to the aquatic insects that dominate the headwater streams. In the case of Crustacea and other wholly aquatic groups like Mollusca, their evolutionary ancestors moved directly into freshwater environments from marine or estuarine environments (Thorp and Covich, 1991).

Some species, such as *Daphnia magna* (Martinez-Jeronimo and Martinez-Jeronimo, 2007), have populations found in brackish waters. Aladin and Potts (1995) describe Cladocera as strong osmoregulators. Hence, in addition to not being found in these headwater streams, the standard invertebrate test species do not appear to be sensitive to the sorts of major ions leaching from valley fills.

The evolutionary ancestors of insects moved from marine or estuarine environments into terrestrial environments. Then, in turn, the evolutionary ancestors of aquatic insects, such as Ephemeroptera, Plecoptera, Trichoptera, and Odonata moved from terrestrial environments to freshwater environments (Merritt and Cummins, 1996). As a result, aquatic insects possess very different mechanisms for osmotic regulation compared to the wholly aquatic groups. In the aquatic insects found in these streams, osmotic regulation is accomplished in part by tissues called chloride cells or chloride epithelia, which are involved in ion absorption, an important adaptation in the low ionic strength, freshwater habitats where aquatic insects are found (Kominick, 1977). In addition, the insects differ from the test species in that their eggs develop externally, so they are directly exposed to contaminated waters. This suggests that Crustacea are not appropriate surrogates for these aquatic insects in laboratory toxicity tests, particularly those that test the effects of the alterations in water chemistry associated with valley fills.

Even insect species, like *Chironomus tentans* and *Isonychia bicolor*, may not be good surrogates. Aquatic Diptera possess anal papillae, which though different in structure are functionally equivalent to chloride epithelia. In the case of *Isonychia bicolor*, its natural distribution is in larger streams with higher alkalinities and conductivities than those found in the streams affected by MTM-VF (Kondratieff and Voshell, 1984). Also, the bioassays testing *Isonychia bicolor* used relatively large (~ 9 to 14 mm in length), late instar nymphs in 7-day tests, where molting and survival were the only measurement endpoints. Other life stages and measurement endpoints appropriate to the survival of these mayfly populations could be more sensitive to the chronic stresses imposed by the observed changes in water chemistry.

8.3.5. Further Investigate Selenium and Sediments

Aqueous selenium concentrations and concentrations in fish fall within a range that can cause effects on fish and fish-eating birds. Additional analyses, including possibly a study of stream-based food webs, could better define the extent of this problem, and reproductive tests of fish collected in high-selenium streams could better define the nature of the problem. To confirm effects of Se on fish reproduction, fish would be collected from high Se streams and spawned in the laboratory. This would be required because Se acts by bioaccumulation in the females and transfer to the eggs. Little is known about the effects of selenium on stream invertebrates. Analyses of invertebrates from high-selenium streams and reproductive tests could determine whether selenium is contributing to observed effects.

Few data are available concerning the effects of MTM-VF on the chemical quality of sediments in streams below valley fills. While dissolved trace metals in effluent waters below valley fills appear to be low, there is evidence along with geochemical theory that particulate metals should be produced within valley fills and may under some conditions be flushed downstream. Also, there are some metals (i.e., Mn, Ni) whose solubilities are not affected by pH and whose dissolved concentrations maybe somewhat elevated in effluent waters. Therefore, data on sediment concentrations of metals could be used to assess whether sediment contamination may be a concern associated with MTM-VF. Observations could also determine if effects associated with the deposition of particulate metals occur. These effects could be similar to those observed with iron hydroxides in more acidic situations. To completely assess this exposure pathway, such sampling could include measurement of pore water concentrations of the dissolved metals and ammonia or use techniques such as simultaneously extracted metals and acid volatile sulfide.

8.3.6. Quantify Cumulative Effects

Additional studies explicitly designed to quantify the cumulative effects of MTM-VF would help differentiate those effects from the other land uses in the Central Appalachian Coalfields, such as abandoned mines, oil and gas development and residential development. Additional water chemistry sampling, combined with spatial analyses of the number and volume of valley fills, could reveal how specific conductivity and other measures of the dissolved ions increase as the percentage of the watershed in valley fills increases and how export of dissolved ions changes with time after the creation of a valley fill.

Currently, little is known about the cumulative effects of incremental loss of headwater streams and naturally occurring mountain aquifers on the region’s hydrology and water supplies. Given the huge scales at which landscape disturbances above and below ground occur in coal mining areas, groundwater sampling, tracer studies and surface flow-groundwater interaction monitoring, in addition to water quality sampling, may be needed.

8.3.7. Quantify Functionality of Constructed Streams

Finally, although there is a large body of literature on stream restoration ecology in urban and agricultural streams, we found there is a lack of evidence on the biota and ecosystem functioning associated with the constructed sediment and flow control channels on valley fills. If these streams are argued to mitigate the effects of stream burial, the type and degree of mitigation should be quantified.

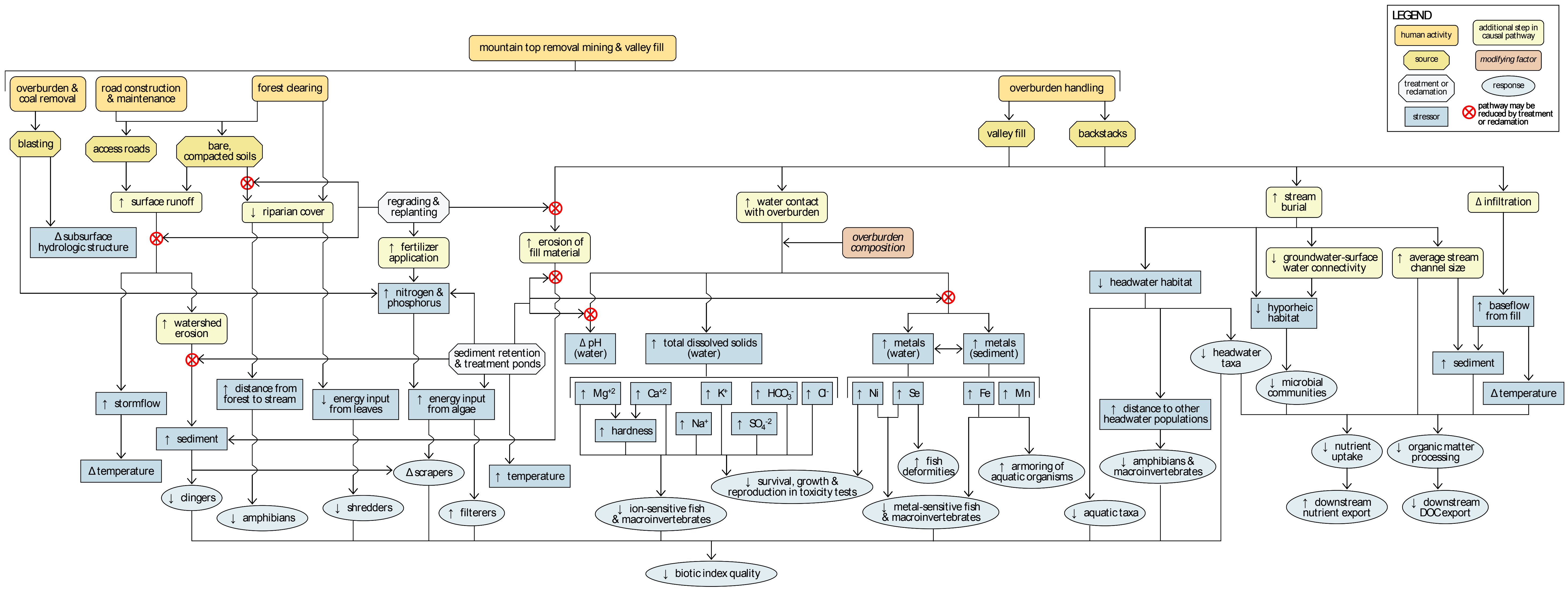

Figure 12. A conceptual model of the impacts of mountaintop mines and valley fills on aquatic ecosystems (narrative description in Section 8.1).

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APPENDIX A
LITERATURE SEARCHES

The peer-reviewed journal articles and reports reviewed in this paper were identified using a variety of search methods. The report authors identified papers using ISI Web of Knowledge℠ and Google™ Scholar and references that either cited, or were cited by the Programmatic Environmental Impact Statement or other relevant papers. This search was supplemented by two more systematic searches described below. Because of the short project timeline, author and supplemental searches were conducted in parallel.

A.1. KEYWORD SEARCH OF ISI WEB OF KNOWLEDGE℠ AND GOOGLE™ SCHOLAR

Publications were identified using ISI Web of Knowledge℠ and Google™ Scholar based on keywords (see Table A-1). The search covered publication dates up to August 2009. ISI Web of Knowledge℠ searches journal articles dating from 1970. Google™ Scholar does not specify a date range but generally sorts the search returns so that more recent references listed first.

The keyword searches of ISI Web of Knowledge℠ and Google™ Scholar produced a total of 277 citations including books, conference proceedings, journal articles, reports, theses/dissertations and other sources (see Table A-2). Note that only the peer-reviewed journal articles and reports were included in our review. Google™ Scholar generally returned more results than ISI Web of Knowledge℠. Google™ Scholar searches the Web across multiple disciplines for journal articles, web documents, government reports, other papers, theses/dissertations, books and abstracts. Searches are performed in such a manner that the most relevant documents appear on the first page. Relevancy is determined by "weighing the full text of each article, the author, the publication in which the article appears and how often the piece has been cited in other scholarly literature." When searching Google™ Scholar, at minimum, the first five pages were checked for relevant papers. Search terms were then refined if necessary. ISI Web of Knowledge℠ returned journal articles that were very specific to the keywords that were entered, which often resulted in fewer or no returns.

Journal articles were classified by region and relevance (see Table A-3). The region of interest was defined as the Central Appalachian Coalfields (see Figure 1). Most of the journal articles judged to be "not relevant" focused on acid mine drainage, rather than the alkaline discharges that are typical of mountaintop mines and valley fills (MTM-VF). Laboratory studies were classified as "stressors in streams from other regions," and a comment was added to indicate that it was a laboratory investigation.

A.2. ECOTOXICOLOGICAL SEARCHES

Searches for ecotoxicological studies on the major ions, and iron, aluminum and manganese were supplemented by keyword and Chemical Abstracts Service (CAS) number searches using BIOSIS, CAS, TOXLINE, Cambridge Scientific Abstracts and U.S. Environmental Protection Agency's (EPA)COTOX reference files.

Of the ecotoxicological searches, the one conducted for sulfate compounds calcium sulfate ($CaSO_4$), magnesium sulfate ($MgSO_4$), potassium persulphate (KSO_4), sodium sulfate ($NaSO_4$), and ferrous sulfate (Fe_xSO_4) was completed in time for inclusion in this appendix. Citations were reviewed for applicability based on criteria such as the subject of the paper, species group studied and analytical methods. Of 1,825 citations identified, 193 were considered to be applicable and relevant to organism groups of interest (see Table A-4). Most of the citations judged to be nonapplicable studied fate and transport rather than effects. The relevant citations were further reviewed for relevance to the ion mixture typically observed in discharges from MTM-VF.

Table A-1. Keywords used for ISI Web of Knowledge^{SM} and Google™ Scholar searches

Keywords			
algae	dissolved oxygen	Mg	sediment transport
alkaline	DO	mine reclamation	sediments
amphibian	electrical conductivity	minnow	selenium
Anuran	Ephemeroptera	mollusc	snail
Appalachian streams	fertilizer	Mollusca	sodium
aquatic biota	fish	mollusk	sodium chloride
aquatic insects	frog	mountain top mining	specific conductance
aquatic toxicity	herpetofauna	mountaintop mining	stoneflies
arsenic	hollow fill	mountaintop removal mining	stonefly
bank stability	hydrologic alteration	mussels	stream temperature
bivalve	leachate	nickel	streams
caddisflies	macroinvertebrate	nutrients	sulfate
caddisfly	macroinvertebrates	overburden	TDS
calcium	macroinvertebrates	periphyton	temperature
coal mine	macrophyte	pH	thermal
coal mine overburden	magnesium	Plecoptera	thermal regime
coal mine spoil	manganese	potassium	toad
conductivity	mayflies	riparian	total dissolved solids
diatom	mayfly	salamander	Trichoptera
discharge	metals	salinity	valley fill

Mg = magnesium; TDS = total dissolved solids.

Table A-2. Breakdown of the literary source of citations from keyword search

Literary source	Number of citations
Books	6
Conference proceedings	22
Journal articles	154
Reports	38
Theses/dissertations	18
Other	39

Table A-3. Categorization of journal articles by region and relevance

Description	Number of citations
MTM-VF in region of interest	5
MTM-VF in other region	0
Stressors in streams of interest	7
Stressors in streams from other regions	59
Review article of stressor of interest	25
Not relevant	30
Defer judgment	46

Table A-4. Breakdown of sulfate ecotoxicological search results by organism group

Organism Group	Number of citations
Fish	62
Herpetofauna	3
Insects	5
Invertebrates	73
Plants	50

APPENDIX B
REGULATORY ISSUES RELATED TO MTM-VF OPERATIONS

Mountaintop mines and valley fills (MTM-VF) operations are permitted by state and federal surface mining and environmental protection authorities. While regulations for individual mines exist under the Surface Mining Control and Reclamation Act, which is implemented by the Office of Surface Mining (OSM) and delegated States with OSM oversight, there are several sections of the Clean Water Act (CWA) that apply. These are implemented by the U.S. Environmental Protection Agency (EPA) and the U.S. Army Corps of Engineers (USACE) and individual states authorized to implement portions of the CWA. A complete listing and interpretation of regulations that affect MTM-VF operations is beyond the scope of this report. The following is a general discussion of how the CWA, and particularly water quality standards, are implemented in the context of MTM-VF.[13]

B.1. IN GENERAL

Section 301 of the CWA prohibits the discharge of any pollutant by any person except in compliance with, *inter alia,* a permit: 33 U.S.C. § 1311(a) "Except in compliance with this section and sections ... 1342 and 1344 of this title, the discharge of any pollutant by any person shall be unlawful." For purposes of MTM-VF, there are two relevant CWA permits. The USACE issues a permit pursuant to Section 404 of the CWA (33 U.S.C. § 1344) for the discharge of dredged and/or fill material. This permit includes construction of the valley fill itself and the fill necessary to create an impounded sediment pond downstream of the toe of the valley fill. The second permit is issued by either the EPA or an authorized State pursuant to Section 402 of the CWA (33 U.S.C. § 1342). The Section 402 program is also known as the "National Pollutant Discharge Elimination System" or "NPDES" program. The NPDES permit includes the discharge from the sediment pond and any stormwater associated with the mining activity.

NPDES permits must include technology-based effluent limitations. For purposes of MTM-VF, the applicable technology-based effluent limitations are set forth at 40 C.F.R. Part 434. In addition to industry sector-specific technology-based effluent limitations,

[13]While beyond the scope of this paper, it is worth noting that the Surface Mining Control and Reclamation Act (SMCRA) and its implementing regulations also state that water quality must be maintained and that water quality standards should not be violated. See, e.g., 30 U.S.C. § 1258(a)(9); 30 U.S.C. § 1265(b)(8)(C); 30 U.S.C. § 1265(b)(10); 30 C.F.R. § 810.2(g); 30 C.F.R. § 816.42; and 30 C.F.R. § 816.57(a)(2). SMCRA also specifically states that it does not supersede the Clean Water Act and other laws related to preserving water quality. See 30 U.S.C. § 1292(a)(3).

Section 301(b)(1)(C) of the CWA requires permits to include limits necessary to achieve water quality standards 33 U.S.C. § 1311(b)(1)(C).

B.2. WATER QUALITY STANDARDS

Water quality standards are the foundation of the water quality-based control program mandated by the CWA. Water quality standards define the goals for a waterbody by designating its uses, setting narrative and numeric criteria to protect those uses and establishing provisions to protect water quality from pollutants. See 40 C.F.R. § 130.3. A water quality standard consists of four basic elements:

(1) **Designated uses** of the water body (e.g., recreation, water supply, aquatic life, agriculture)

(2) **Water quality criteria** to protect designated uses (numeric pollutant concentrations and narrative requirements)

(3) An **antidegradation policy** to maintain and protect existing uses and high quality waters and

(4) **General policies** addressing implementation issues (e.g., low flows, variances, mixing zones).

B.2.1. Designated Uses

The water quality standards regulation requires that States and authorized Indian Tribes specify appropriate water uses to be achieved and protected. Appropriate uses are identified by taking into consideration the use and value of the water body for public water supply, for protection of fish, shellfish and wildlife, and for recreational, agricultural, industrial, and navigational purposes. In designating uses for a water body, States and Tribes examine the suitability of a water body for the uses based on the physical, chemical, and biological characteristics of the water body, its geographical setting and scenic qualities and economic considerations. Each water body does not necessarily require a unique set of uses. Instead, the characteristics necessary to support a use can be identified so that water bodies having those characteristics can be grouped together as supporting particular uses.

West Virginia has designated all waters of the state with an aquatic life use (ALU):

> **§47-2-6. Water Use Categories**.
> 6.1. These rules establish general Water Use Categories and Water Quality Standards for the waters of the State. Unless otherwise designated by these rules, at a minimum, all waters of the State are designated for the Propagation and Maintenance of Fish and Other Aquatic Life (Category B) and for Water Contact

Recreation (Category C) consistent with Federal Act goals. Incidental utilization for whatever purpose may or may not constitute a justification for assignment of a water use category to a particular stream segment.

In addition, West Virginia Department of Environmental Protection (WVDEP) water quality standards specify that waste assimilation and transport are not recognized as a designated use:

§47-2-6. Water Use Categories.
6.1.a. Waste assimilation and transport are not recognized as designated uses. The classification of the waters must take into consideration the use and value of water for public water supplies, protection and propagation of fish, shellfish and wildlife, recreation in and on the water, agricultural, industrial and other purposes including navigation.

B.2.2. Water Quality Criteria

States establish criteria necessary to protect the designated use. Water quality criteria may take the form of either specific numeric criteria, such as concentrations of a particular pollutant, or narrative description of water quality conditions.

B.2.2.1. *Numeric Criteria*

Section 304(a)(1) of the Clean Water Act requires us to develop numeric criteria for water quality that accurately reflect the latest scientific knowledge. These criteria are based solely on data and scientific judgments on pollutant concentrations and ecological or human health effects. Section 304(a) also provides guidance to States and Tribes in adopting water quality standards. Numeric criteria are developed for the protection of aquatic life as well as for human health.

Numeric aquatic life criteria are generally pollutant-specific and reflect numeric limits on the amount of a pollutant that can be present in a water body without harm to indigenous aquatic life. Aquatic life criteria are designed to provide protection for aquatic organisms from the effects of acute (short-term) and chronic (long-term) exposure to potentially harmful chemicals.

Human health criteria set allowable concentrations based on human exposure to water pollutants when humans drink untreated surface water or eat fish, shellfish, or wildlife that have been contaminated by pollutants in surface waters. To reduce the risk to humans from these sources, EPA scientists research information to determine the levels at which specific chemicals are not likely to adversely affect human health.

In making water quality management decisions, a State or Tribe should independently apply each criterion that has been adopted into its water quality standards. If a water body has multiple designated uses with different criteria for the same pollutant, States/Tribes should use the criterion protective of the most sensitive use.

B.2.2.2. *Narrative Criteria*

While numeric criteria help protect a water body from the effects of specific chemicals, narrative criteria protect a water body from the effects of pollutants that are not easily measured, or for pollutants that do not yet have numeric criteria, such as chemical mixtures, suspended and bedded sediments and floatable debris.

West Virginia's narrative water quality criteria are set forth in a portion of the West Virginia regulations known as "Conditions Not Allowed:"

> **WV §47-2-3. Conditions Not Allowable In State Waters.**
> 3.2.i. Any other condition, including radiological exposure, which adversely alters the integrity of the waters of the State including wetlands; no significant adverse impact to the chemical, physical, hydrologic, or biological components of aquatic ecosystems shall be allowed.

Other examples presented here include excerpts from Kentucky surface water standards (Chapter 10); the narrative standards in 401 KAR 10:026-031, which state in part

> **001 Definitions 401 KAR Chapter 10**
>
> (5) "Adversely affect" or "adversely change" means to alter or change the community structure or function, to reduce the number or proportion of sensitive species, or to increase the number or proportion of pollution tolerant aquatic species so that aquatic life use support or aquatic habitat is impaired.
>
> (38) "Impact" means a change in the chemical, physical, or biological quality or condition of surface water.
>
> (39) "Impairment" means, a detrimental impact to surface water that prevents attainment of a designated use.

> **401 KAR 10:031, Section 2: Minimum Criteria Applicable to All Surface Waters.**
>
> (1) The following minimum water quality criteria shall be applicable to all surface waters including mixing zones, with the exception that toxicity to aquatic life in mixing zones shall be subject to the provisions of 401 KAR 10:029, Section 4. Surface waters shall not be aesthetically or otherwise degraded by substances that

(a) Settle to form objectionable deposits;

(b) Float as debris, scum, oil, or other matter to form a nuisance;

(c) Produce objectionable color, odor, taste, or turbidity;

(d) Injure, are chronically or acutely toxic to or produce adverse physiological or behavioral responses in humans, animals, fish and other aquatic life;

(e) Produce undesirable aquatic life or result in the dominance of nuisance species;

(f) Cause fish flesh tainting.

A narrative chemical criterion for total dissolved solids and specific conductance reads

401 KAR 10:031, Section 4: Aquatic Life.
(f) Total dissolved solids or specific conductance. Total dissolved solids or specific conductance shall not be changed to the extent that the indigenous aquatic community is adversely affected.

B.2.2.2. *Establishing Impairment*

Section 303(d) of the CWA requires States to periodically identify those waters that are not expected to achieve water quality standards even after application of technology-based effluent limitations to NPDES-permitted point sources (33 U.S.C. § 1313(d)). This identification is commonly referred to as the State's "Section 303(d) list." By regulation, States must submit their Section 303(d) lists to EPA for approval every even-numbered year (40 C.F.R. § 130.7(d)). In establishing its Section 303(d) list, States must consider all existing and readily available information, including predictive models (40 C.F.R. § 130.7(b)(5)).

In July 1991, EPA transmitted final national policy on the integration of biological, chemical and toxicological data in water quality assessments. According to this policy, referred to as "Independent Application," indication of impairment of water quality standards by any one of the three types of monitoring data (biological, chemical, or toxicological) should be taken as evidence of impairment regardless of the findings of the other types of data. This policy continues to the present. See, e.g., *Guidance for 2006 Assessment, Listing and Reporting Requirements Pursuant to Sections 303(d), 305(b) and 314 of the Clean Water Act*.

EPA supports use of biological assessments as a direct measure of whether the water body is achieving the designated use and relevant narrative criteria. A water body in its natural condition is free from the harmful effects of pollution, habitat loss and other negative stressors. It is characterized by a particular biological diversity and abundance of organisms. This biological integrity—or natural structure and function of aquatic life—can be dramatically different in various types of water bodies in different parts of the country. EPA recognizes that biological assessments are a direct measure of the aquatic life use. Because of the natural variability in ecosystems and aquatic life around the country, EPA could not develop national biocriteria. Instead, EPA developed methodologies that States can use to assess the biological integrity of their waters and, in so doing, set protective water quality standards. These methodologies describe scientific methods for determining a particular aquatic community's health and for maintaining optimal conditions in various bodies of water. States use these standard methods to develop their own bioassessment methods and tools. Bioassessment results are used to support many programs under the CWA (see Figure B-1).

The States have increasingly relied upon biological monitoring in lieu of ambient water chemistry monitoring because biological monitoring allows the States to maximize monitoring resources and to assess a larger percentage of their waters. Since 2004, West Virginia has utilized standard field collection, laboratory and data analysis methods to use its biological assessment data. This has resulted in West Virginia's use of a family-level benthic metric developed jointly by EPA and West Virginia Department of Environmental Protection called the West Virginia Stream Condition Index (WV SCI) to identify impairment of the aquatic life use. See http://www.wvdep.org/Docs/536_WV-Index.pdf. West Virginia also developed an assessment methodology for using the WV SCI to interpret its narrative criterion and to make aquatic life use-attainment decisions. For an example, see WVDEP's 2008 Integrated Water Quality Monitoring and Assessment Report available at http://www.wvdep.org/Docs/16495_WV_2008_IR_Supplements_Complete_Version_EPA_Approved.pdf.

In Kentucky, the Kentucky Division of Water assessment methodologies for ALU attainment are similar, where the State uses biological monitoring data and statistical-based multimetric index analyses to assess waterbody attainment. For macroinvertebrates, the KY Macroinvertebrate Bioassessment Index is used to evaluate ALU: http://www.water.ky.gov/NR/rdonlyres/7F189804-4322-4C3E-B267-5A58E48AAD3F/0/Statewide_MBI.pdf.

In nonheadwater streams, KY uses fish communities as other indicators of ALU with the KY Index of Biotic integrity, a similarly constructed multimetric index: http://www.water.ky.gov/NR/rdonlyres/04C65101-AF1C-4751-809B-4F5D09B7269A/0/KIBI_paper.pdf.

Section 303(d) also requires the States to establish total maximum daily loads (TMDLs) for their impaired waters. Essentially, a TMDL is a measure of the assimilative capacity of a waterbody considering seasonal variability and critical conditions, allocated among point sources and nonpoint sources and incorporating a margin of safety. See 33 U.S.C. § 1313(d); 40 C.F.R. § 130.2(i); and130.7(c).

B.3. IMPLEMENTATION OF WATER QUALITY STANDARDS THROUGH NPDES PERMITS

As set forth above, Section 301 of the CWA requires NPDES permits to contain both technology-based effluent limitations and water quality-based effluent limitations. For the industry sector, that includes surface coal mining with valley fills, the applicable technology-based effluent limitations are set forth at 40 C.F.R. Part 434. These effluent limitations include limitations on discharges from coal preparation plants, acid and alkaline mine drainage, postmining areas, remining and western alkaline mining. For example, effluent limitations on discharges from a new source of alkaline mine drainage include limits on iron, total suspended solids and pH. See 40 C.F.R. § 434.45.

The NPDES regulations implement the water quality-based effluent limitations requirement as set forth in CWA Section 301(b)(1)(C) through the following regulatory requirements:

> No permit may be issued ... (d) When the imposition of conditions cannot ensure compliance with the applicable water quality requirements of all affected States… (40 C.F.R. § 122.4(d)).
>
> [E]ach NPDES permit shall include conditions meeting the following requirements when applicable .... [A]ny requirements in addition to or more stringent than promulgated effluent limitations guidelines ... necessary to: achieve water quality standards under Section 303 of the CWA, including State narrative criteria for water quality… (40 C.F.R. § 122.44(d)(1)).
>
> No permit may be issued ... (i) To a new source or a new discharger, if the discharge from its construction or operation will cause or contribute to the violation of water quality standards (40 C.F.R. § 122.4(i)).

Most States, including West Virginia and Kentucky, have been authorized to issue NPDES permits for discharges to waters within their borders. EPA retains the ability to review, object to and if necessary, take over issuance of a particular NPDES permit. See 33 U.S.C. § 1342(d); 40 C.F.R. § 123.44. The scope of EPA's NPDES permit review in a particular state is generally spelled out in a memorandum of agreement with that state (40 C.F.R. § 123.44). EPA also retains the ability to enforce discharges without or in violation of an NPDES permit (33 U.S.C. § 1319).

B.4. IMPLEMENTATION OF WATER QUALITY STANDARDS THROUGH SECTION 404 PERMITS

Section 404(b)(1) directs the EPA in conjunction with the Secretary of the Army to establish guidelines to be applied by the USACE when considering an application for a permit to discharge dredged and/or fill material pursuant to Section 404 of the CWA.. This instruction has resulted in the Section 404(b)(1) Guidelines (40 C.F.R. Part 230), which provide the substantive environmental criteria that must be applied by the USACE when considering a Section 404 permit application. Among other things, the USACE may issue a permit only if it determines that the project represents the least damaging practicable alternative:

> [N]o discharge of dredged or fill material shall be permitted if there is a practicable alternative to the proposed discharge which would have less adverse impact on the aquatic ecosystem, so long as the alternative does not have other significant adverse environmental consequences (40 C.F.R. § 230.10(a)).

The USACE also must ensure that the project proponent has taken "all appropriate and practicable steps to avoid and minimize adverse impacts to waters of the United States" (33 C.F.R. § 332.1(c)); see also 40 C.F.R. § 230.10(a)(1)(i); 40 C.F.R. § 230.10(d); and 40 C.F.R. §§ 230.70-.77.

In addition, the Section 404(b)(1) Guidelines prohibit the issuance of a permit "if it: (1) Causes or contributes, after consideration of disposal site dilution and dispersion, to violations of any applicable State water quality standard," (40 C.F.R. § 230.10(b)(1)), or if it "will cause or contribute to significant degradation of the waters of the United States, ... [including] (1) Significantly adverse effects of the discharge of pollutants on human health or welfare, including but not limited to effects on municipal water supplies, plankton, fish, shellfish, wildlife and special aquatic sites. (2) Significantly adverse effects of the discharge of pollutants on life stages of aquatic life and other wildlife dependent on aquatic ecosystems, including the transfer, concentration and spread of pollutants or their byproducts outside of the disposal site through biological, physical and chemical processes; (3) Significantly adverse effects of the discharge of pollutants on aquatic ecosystem diversity, productivity and stability...." (40 C.F.R. § 230.10(c)). The USACE also must consider the effect of the discharge on fish, crustaceans, mollusks and other aquatic organisms in the food web (40 C.F.R. § 230.31), the effect on benthos (40 C.F.R. § 230.61(b)(3)) and the suitability of water bodies for populations of aquatic organisms (40 C.F.R. § 230.22).

Before issuing a federal permit or license, federal agencies, including the USACE, must obtain a certification from the State in which the discharge will originate that the discharge will comply with applicable provisions of 33 U.S.C. § 1311, 1312, 1313, 1316 and 1317. Among other things, therefore, the USACE must obtain a certification that the discharge will comply with applicable water quality standards, which are established pursuant to 33 U.S.C. § 1313. In considering the potential of a discharge to cause or contribute to an excursion from water quality standards, the USACE generally will consider conclusive the State's CWA Section 401 water quality certification, unless EPA advises of other water quality aspects to be taken into consideration (33 C.F.R. § 320.4(d)).

While the USACE is the permit-issuing authority for Section 404, EPA retains significant authorities, including the authority to prohibit, deny or restrict the use of any defined area for specification as a disposal site pursuant to Section 404(c) (33 U.S.C. § 1344(c)), the ability to request consideration of particular permits by the USACE at the Headquarters level pursuant to the Memorandum of Agreement described in Section 404(q) (33 U.S.C. § 1344(q)), the ability to identify waters that are within the scope of the CWA and to determine the applicability of exemptions pursuant to a Memorandum of Agreement with the USACE under Section 404(f) (33 U.S.C. § 1344(f)), and the ability to enforce discharges without a permit (33 U.S.C. § 1319).

Figure B-2 depicts the sequence of actions necessary to address impaired streams under the CWA.

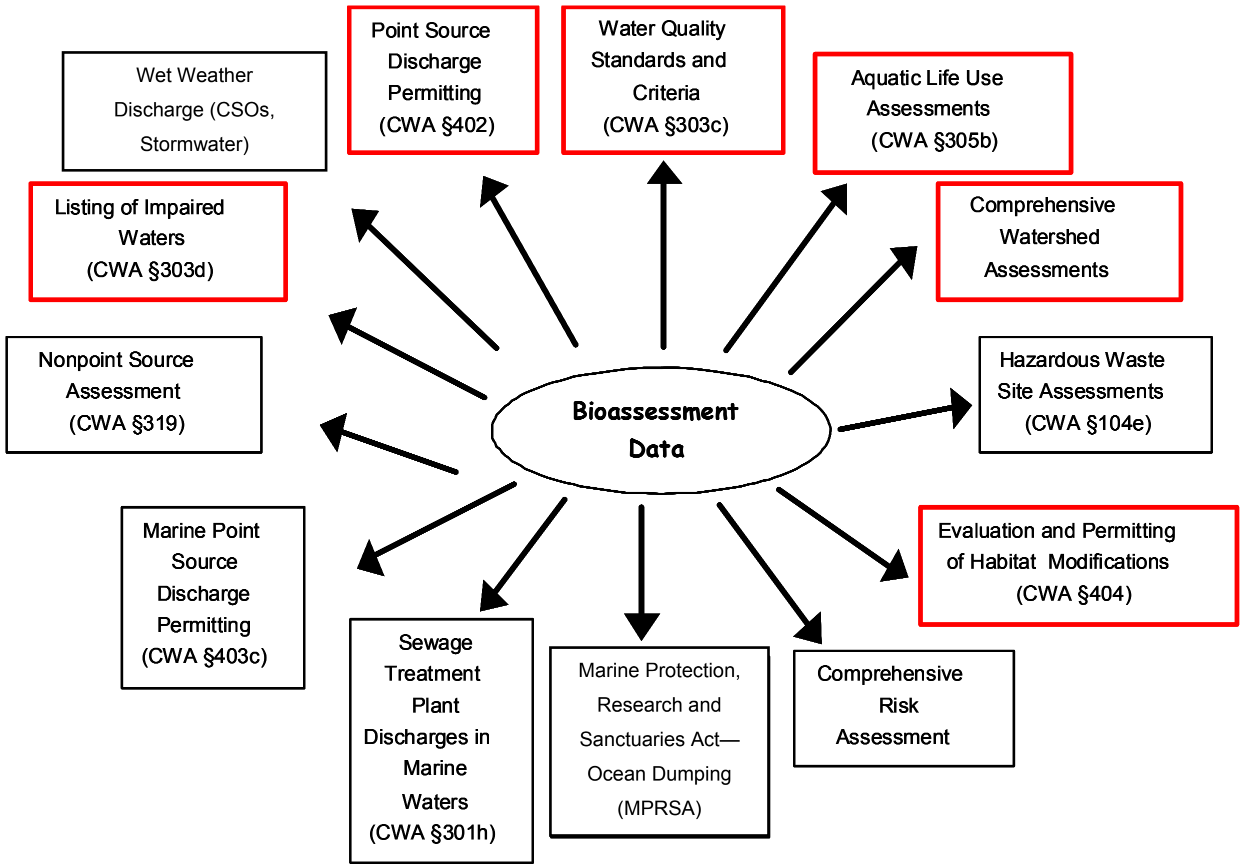

Figure B-1. Simple representation of CWA programs that rely on biological assessment data for program implementation. Coal mining activities sections highlighted in red.

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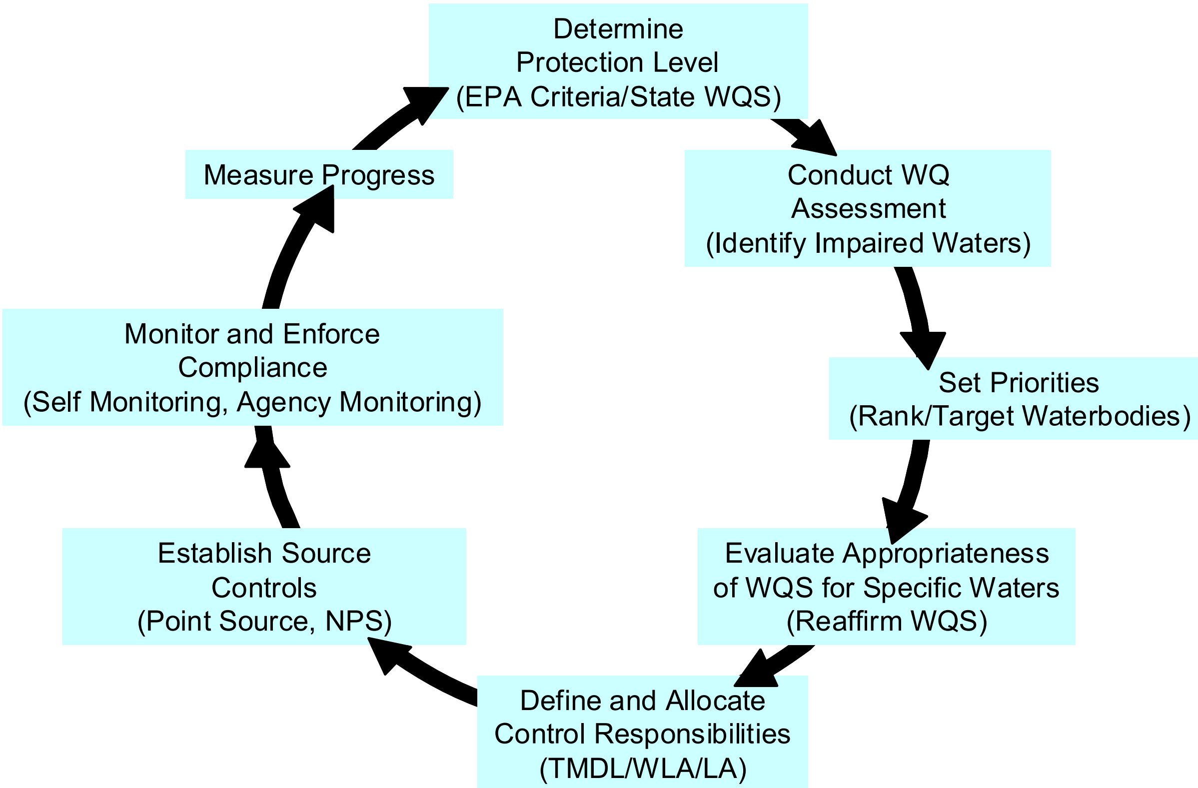

Figure B-2. Water quality-based approach to pollution control.