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

United States Environmental Protection Agency
Office of Research and Development, National Center for Environmental Assessment

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National Center for Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC 20460

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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, and ephemeral, intermittent, 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 degraded.

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CONTENTS

CONTENTS (continued)

CONTENTS (continued)

LIST OF TABLES

LIST OF FIGURES

LIST OF FIGURES (continued)

LIST OF ABBREVIATIONS AND ACRONYMS

AWQC	ambient water quality criterion
BCG	biological condition gradient
$CaCO_3$	calcite
$CaMg(CO_3)$	dolomite
CWA	Clean Water Act
EC_x	effect concentration for x% of the tested organisms
EPA	U.S. Environmental Protection Agency
EPT	Ephemeroptera, Plecoptera, and Trichoptera
FEC_x	field-based effect concentration for x% of the tested organisms
FRA	Forestry Reclamation Approach
GIS	geographic information system
GLIMPSS	genus-level index of most probable stream status
HBI	Hilsenhoff Biotic Index
HCO_3^-	bicarbonate
IBI	Index of Biotic Integrity
KSO_4	potassium persulphate
LC_x	lethal concentration for x% 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
$NaHCO_3$	sodium bicarbonate
NPDES	National Pollutant Discharge Elimination System
OSMRE	Office of Surface Mining Reclamation and Enforcement
PAH	polycyclic aromatic hydrocarbons
PEIS	Programmatic Environmental Impact Statement
SMCRA	Surface Mining Control and Reclamation Act
TDS	total dissolved solids
USACE	United States Army Corps of Engineers
WVDEP	West Virginia Department of Environmental Protection
WVSCI	West Virginia Stream Condition Index

FOREWORD

Headwater streams and watersheds in Appalachia are keystone components of 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 biodiversity. 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 the 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 (reviewed herein) 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 Reclamation and Enforcement (OSMRE), the U.S. 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 report was externally peer reviewed by EPA's Science Advisory Board (SAB) and reflects the SAB's comments and suggestions. In addition, comments received from the public, the mining companies, and environmental groups were evaluated in preparing this final report. This final peer-reviewed assessment will inform the EPA as the Agency continues to implement its regulatory responsibilities under the Clean Water Act.

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PREFACE

This assessment of the effects of mountaintop mines and valley fills on aquatic ecosystems was requested by the U.S. Environmental Protection Agency's (EPA's) Office of Water and regional offices. It will be used to inform the EPA's reexamination of its reviews of Appalachian surface coal mining operations under the Clean Water Act, the National Environmental Policy Act, and the Environmental Justice Executive Order (E.O. 12898). The report was prepared by the National Center for Environmental Assessment in EPA's Office of Research and Development.

The assessment reviews and evaluates evidence from peer-reviewed sources published up through December 2010 and the Programmatic Environmental Impact Statement and its associated appendices published in 2003 and 2005. The external review draft released April 2010 (EPA/600/R-09/138A) was reviewed by EPA staff and panel of the EPA's Science Advisory Board (SAB) that convened July 20 to 22, 2010 (EPA-SAB-11-005, available online at www.epa.gov/sab). In addition, hundreds of comments from the mining companies, other government agencies, nonprofit environmental and scientific organizations, and private citizens were received through the docket or at the SAB panel meeting. Comments from all of these sources were considered and used to improve the clarity and scientific rigor of the document.

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1. EXECUTIVE SUMMARY

Mountaintop mines and valley fills (MTM-VF) lead directly to five principal alterations to stream ecosystems (see Figure 1): (1) springs, and ephemeral, 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 and observed in the effluent waters below valley fills. Downstream ion concentrations were accurately predicted using a simple dilution model, indicating that concentrations decrease primarily when diluted by a cleaner source of water—for example, an unmined tributary. 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” or “biologically impaired” at sites downstream of MTM-VF. Declines in macroinvertebrate indices were observed at ion concentrations well below those associated with effects in tests of mine effluent using standard laboratory organisms.

Selenium concentrations are also elevated downstream of MTM-VF. Selenium can bioaccumulate through aquatic food webs—especially in ponds and reservoirs where retention is high and food webs are long. 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-duration Ambient Water Quality Criterion (AWQC) for selenium. Selenium has been associated with

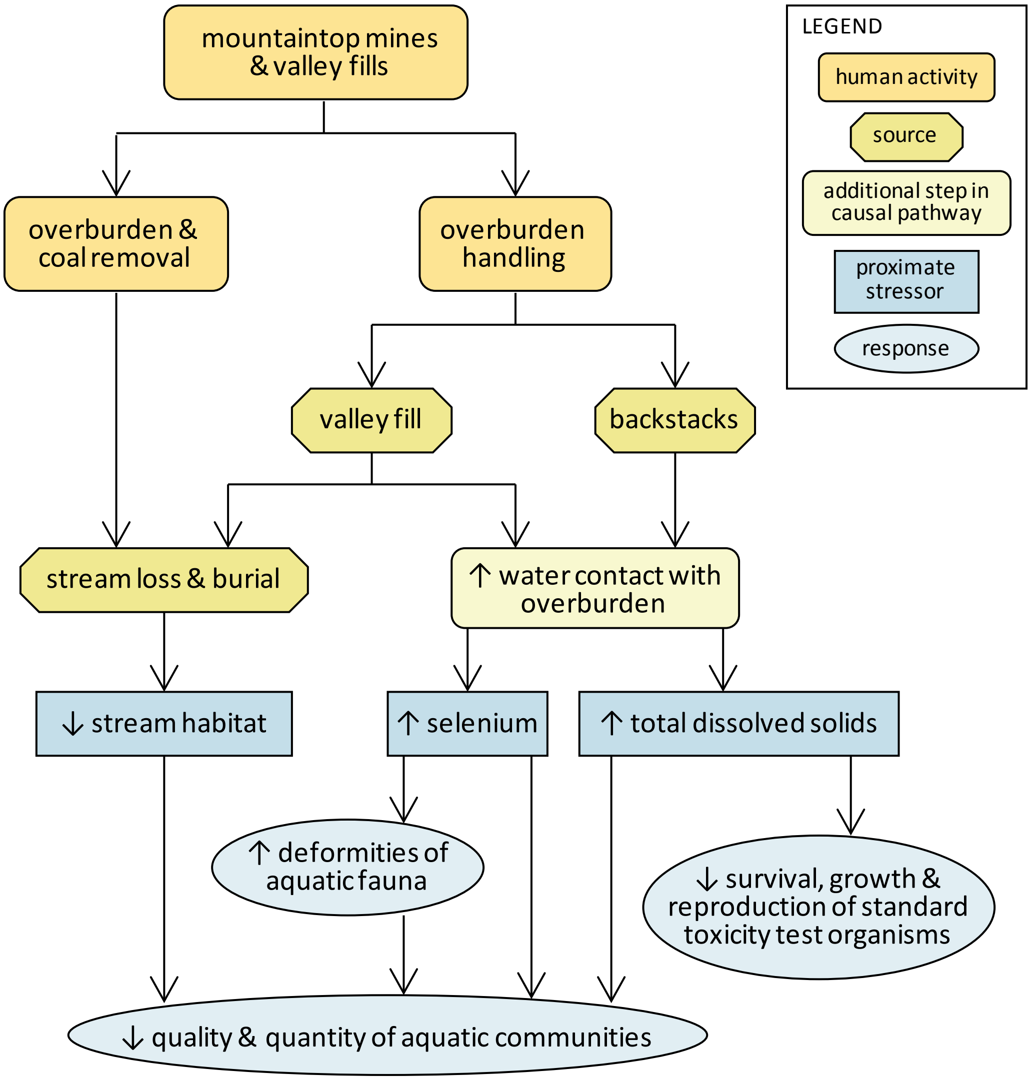

Figure 1. A summary diagram of the principal observed and expected effects of MTM-VF on aquatic ecosystems.

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 2% 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 over 2002 levels by 2012. An updated inventory that would support statistically sound estimates of cumulative stream loss 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 past and 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] As defined in the PEIS, the coalfields cover about 48,000 square kilometers (12 million acres) in West Virginia, Kentucky, Virginia and Tennessee, USA (see Figure 2) (U.S. EPA, 2003, 2005).

The Central Appalachian Coalfields have a long history of mining. Current mining practices, including MTM-VF, employ methods to control the acid mine drainages 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 reexamination 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 (see Figure 3):

- Loss of headwater 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)
- Cumulative impacts of multiple mining operations (see subsections of Sections 3, 4, and 6)
- Effectiveness of on-site reclamation and mitigation activities (see Section 7)

We reviewed the impacts on terrestrial ecosystems from the narrow perspective of their effects on aquatic ecosystems. Our review of reclamation and mitigation practices was limited to their effectiveness in improving on-site aquatic ecosystems. We did not evaluate the impacts of MTM-VF on cultural or aesthetic resources, or human health.

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

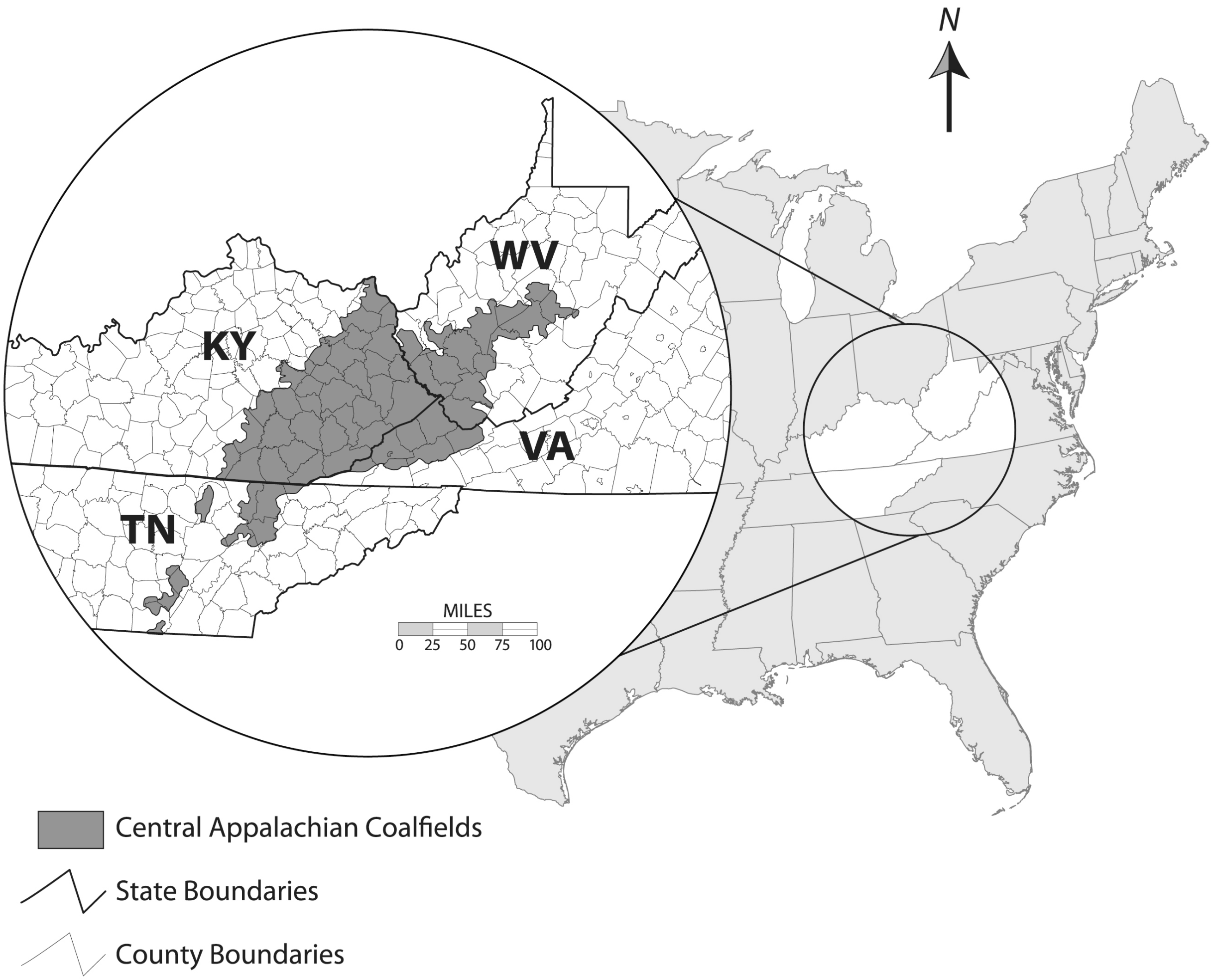

Figure 2. The central Appalachian coalfields.

Source: EPA (2003, 2005).

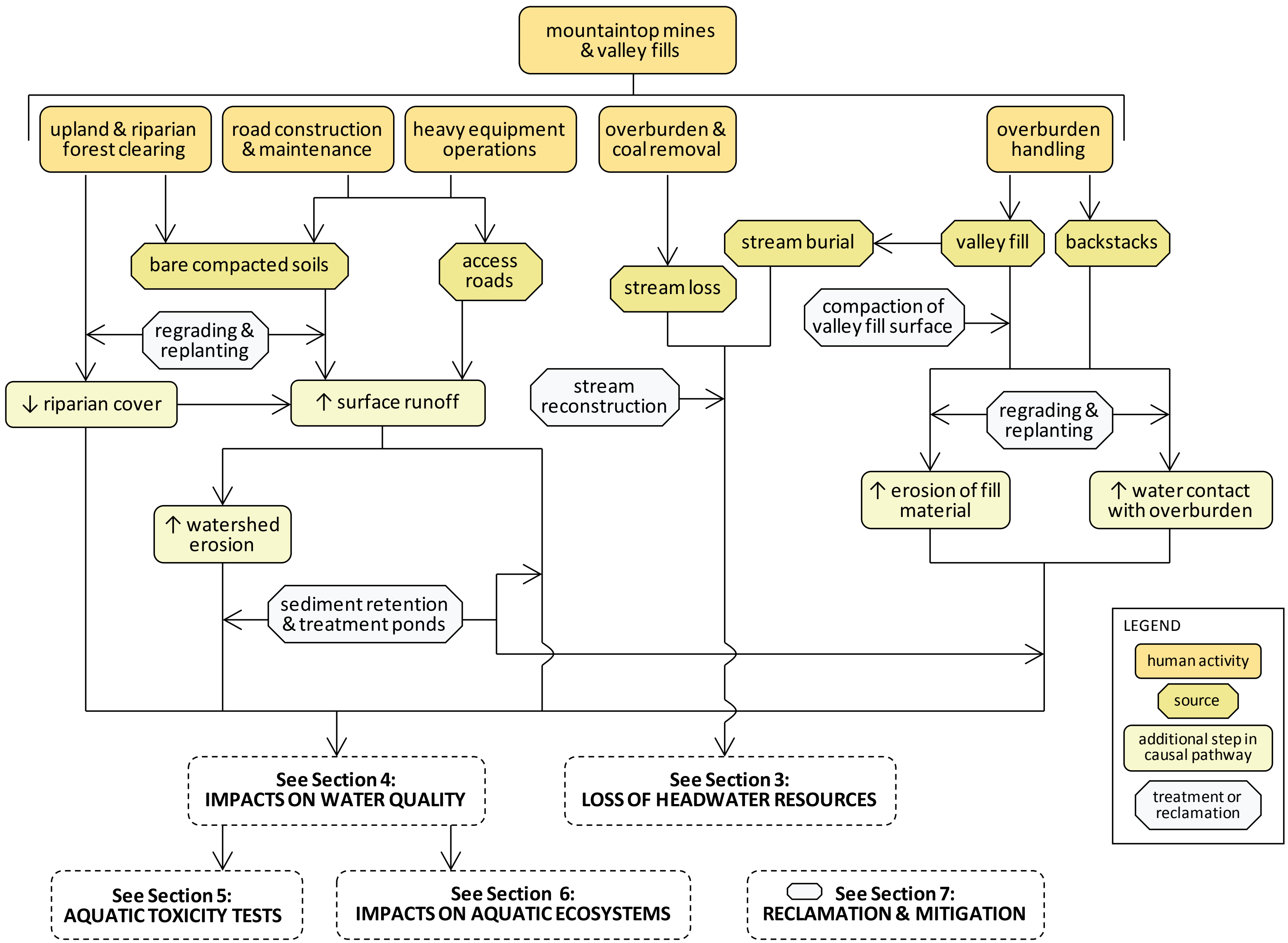

Figure 3. An overview of activities and sources associated with MTM-VF.

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 (see 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 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 textbooks 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. This overview of the processes used in MTM-VF summarizes the description in the PEIS (U.S. EPA, 2003, 2005; see Figure 3). 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 4). As much as 300 vertical meters (1,000 ft) of overburden are removed. With it, any springs, ephemeral, intermittent, and small perennial streams on the mountain's surface are also 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 stored on the mined surface in backstacks and used to recontour the surface. The excess overburden is disposed of in constructed fills in valleys or hollows adjacent to the mined site. These fills bury additional springs and ephemeral, intermittent and small perennial streams.

Both water flow and sediment discharges are altered by MTM-VF (see Figure 5). The heavy equipment used to mine and move the overburden compacts the bare soils, forming a large, relatively impervious surface on the mined site that increases surface runoff. 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

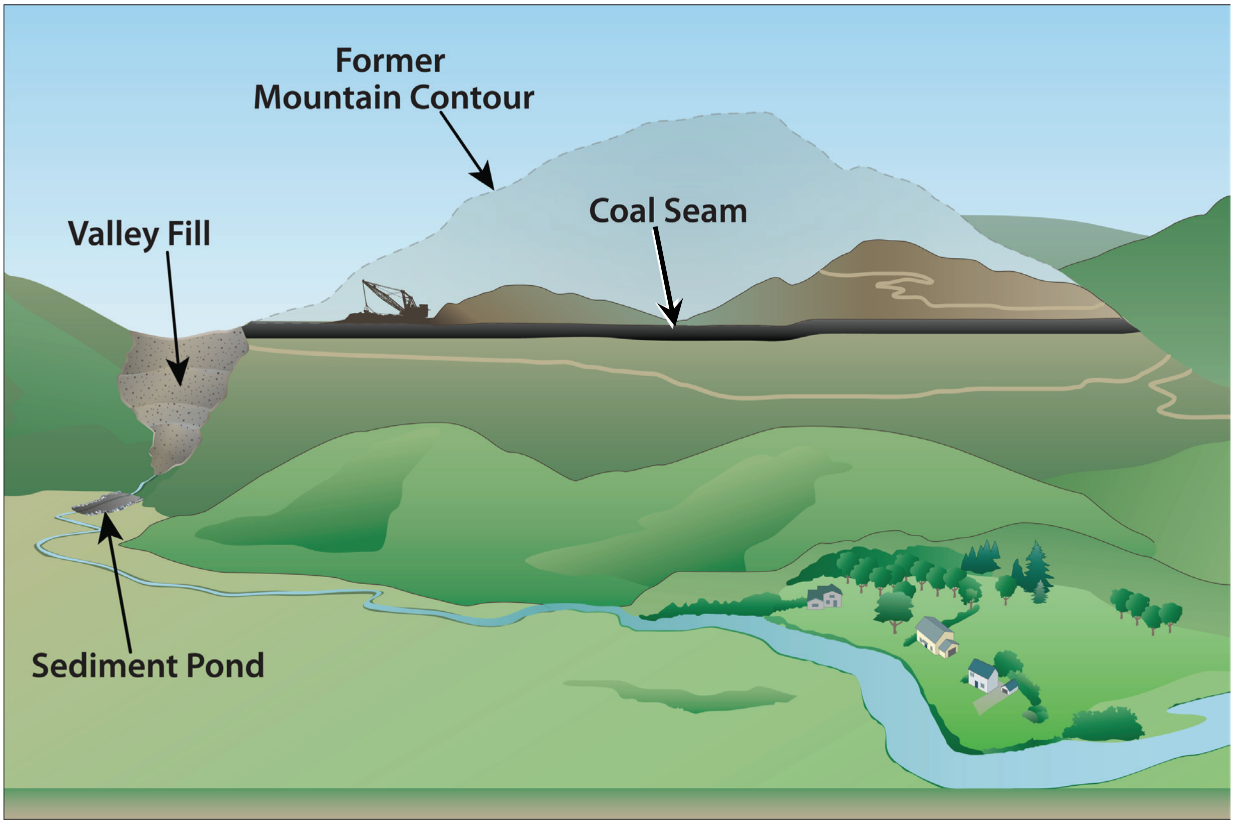

Figure 4. A watershed view of a mountaintop mine and valley fill (no consistent scale).

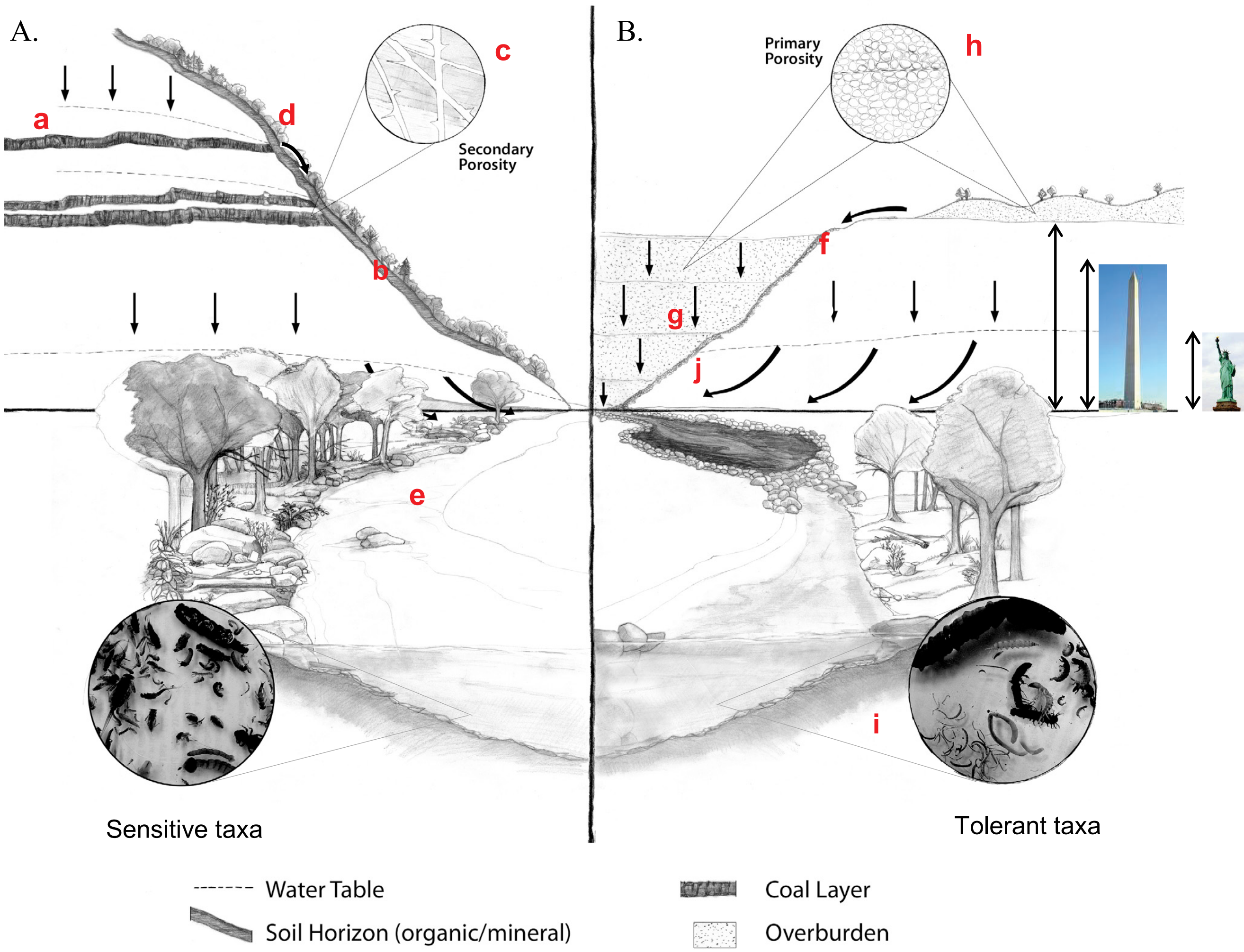

Figure 5. Small stream watershed before and after mountaintop mining and creation of a valley fill (simplified view). Monuments added for scale. Scales differ between upper and lower halves of diagram.

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 local aquifers (a) soil layer interflows (b) and minute stress fractures in geologic strata of the parent mountain (c). Overland flow and subsurface flows (indicated by arrows) form seeps or springs (d) and channelized flows (e) that integrate features of the entire landscape, including riparian vegetation and diverse, in-stream 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 (f) and valley fill (g) height is approximate). Infiltration though valley fills of water exposed to larger total surface area of porous unweathered rock (h) produces higher channelized flows and higher concentrations of dissolved ions and trace metals downstream, where biological communities shift towards tolerant taxa (i). Subsurface flowpaths in the intact geologic strata vary, depending on the types of rock in them, but water tables can ‘back up’ against the valley fill, elevating the water level in the fill, as shown here (j), increasing baseflows and exposure to valley fill materials.

Photographs of macroinvertebrates by Greg Pond.

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 can 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 6) and can 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 fills, valley fills from mountaintop mining operations are expected to be larger because of the volume of material involved. 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 7). 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 3.4% of the land cover in the central Appalachian coalfields. As of 2001, permits for 6,697 valley fills were approved (U.S. EPA, 2002).

Surface mining and reclamation have been identified as the dominant driver of land cover/land-use change in the central Appalachian coalfields and have produced significant changes in the region's topography, hydrology, vegetation, groundwater, and wildlife (Townsend et al., 2009; Loveland et al., 2003; U.S.EPA, 2003, 2005). Coal mining in this region was identified as the greatest contributor to earth-moving activity in the United States (Hooke, 1999; see Figure 8).

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 Reclamation and Enforcement (OSMRE) and by delegated states under OSMRE 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.

A.

B

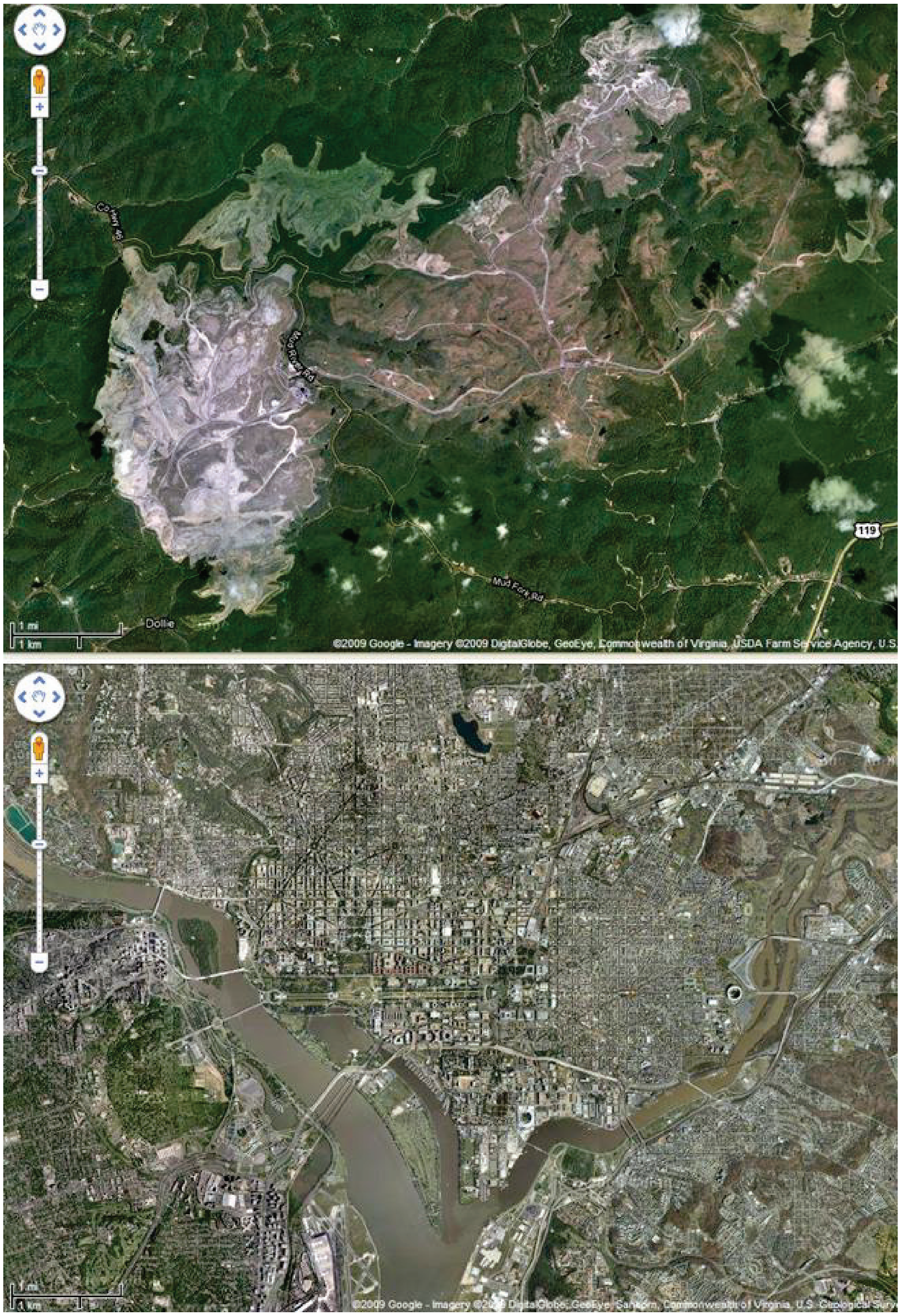

Figure 6. Satellite images of the 40-km^2 Hobet 21 mine (Boone County, WV) (Panel A), and the Washington DC area (Panel B), at the same scale.

Source: Google Maps (2009).

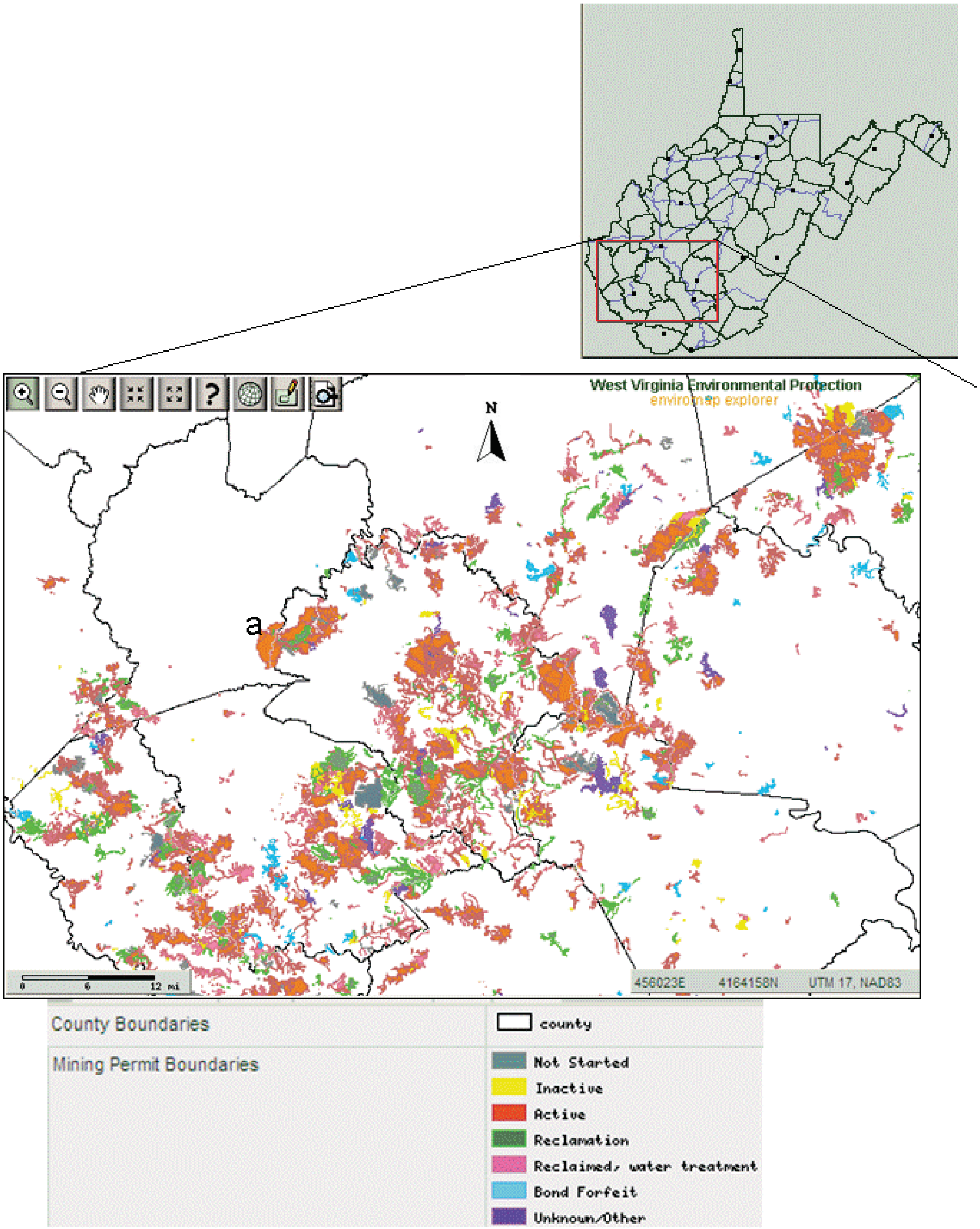

Figure 7. Permit boundaries for surface and underground mines in southwestern West Virginia. The Hobet 21 mine is shown in middle left near Point a.

Source: WVDEP (2009a). Colors modified to improve legibility.

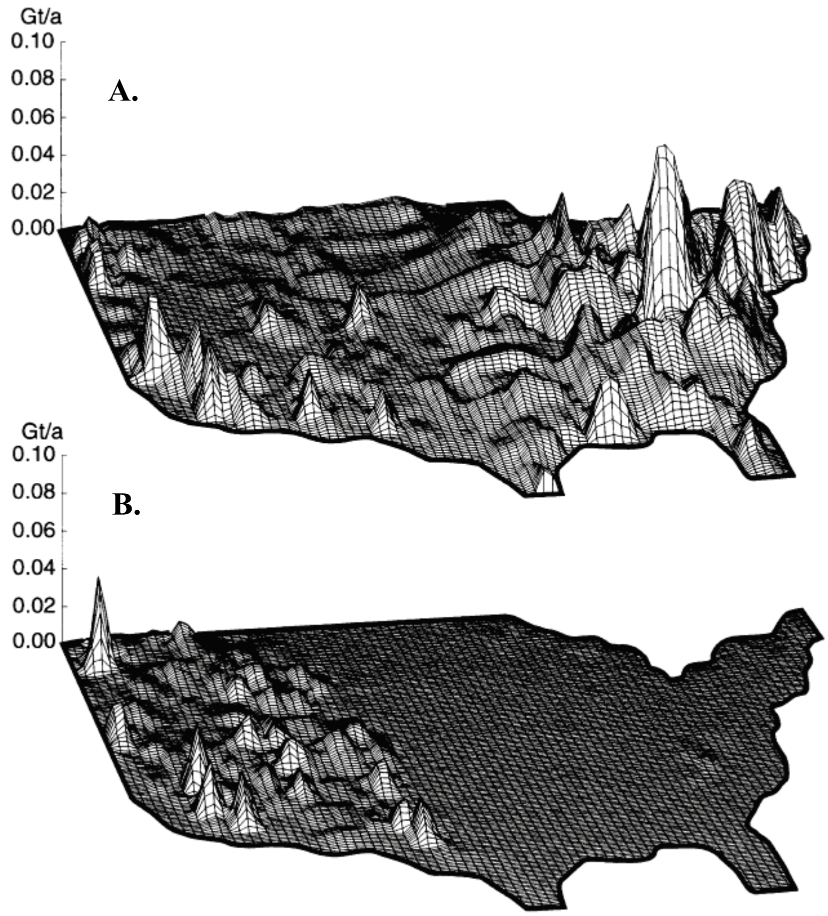

Figure 8. Earth movement by humans and streams. Maps of the United States showing, by variations in peak height, the rates at which earth is moved in gigatonnes per annum in a grid cell measuring 1° (latitude and longitude) on a side, by (A) humans and (B) rivers.

Source: Hooke (1999), used with permission from the publisher.

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, hydrologic, 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)).
- From Kentucky: *"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)).

3. LOSS OF HEADWATER 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 on the removed mountaintops; burial of streams in the actual footprint of the valley fills; and potential fragmentation of remaining stream and riparian habitats.

3.1. BACKGROUND

The term "headwaters" refers to the numerous small channels that form the origins of a stream or river network. Headwaters are characterized by small drainage area, shallow channels, and variable flow. They include hillside springs and seeps, creeks permanently or seasonally connected to local or regional groundwater sources, and transitional channels that flow only during periods of rainfall or snowmelt. Variation in the timing and duration of flow, and the relative contributions of groundwater and stormwater inputs are used to classify headwater streams as *perennial*, *intermittent*, or *ephemeral* (Hewlett, 1982). Perennial headwaters are predominantly groundwater-fed and have continuous surface or subsurface flow except in exceptionally dry periods; intermittent streams flow seasonally (e.g., 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). A single stream channel can have reaches in all three flow duration classes.

An alternative way to classify headwater streams is by stream order (Strahler, 1957). Streams without tributaries are first-order streams, second-order streams are formed when two first-order streams join, third-order streams are formed when two second-order streams join, and so forth. First- and second-order streams are typically classified as headwaters (Meyer and Wallace, 2001; Gomi et al., 2002; Benda et al., 2005). Because stream order is usually determined by using maps, determinations will vary with the scale and accuracy of the map (Leopold, 1994).

The hydrology and setting of headwaters influence their function, especially the transformation and transport of water, organic matter, sediment, and other materials downstream (Paybins, 2003; Freeman et al., 2007; Nadeau and Rains, 2007). Flow properties are influenced by drainage area, climate, topography, channel morphology, underlying geology, and other local and regional factors. Field classification of headwater streams by stream order or flow duration class (ephemeral, intermittent, or perennial) requires an understanding of the geographic setting (Paybins, 2003; Fritz et al., 2006).

In the central Appalachian region, natural headwaters are forested, high-gradient streams that occur on hilltops and in intervening valleys. They are typically dendritic (branched pattern similar to tree roots) in structure with channels confined by underlying rock layers. In unmined areas, hilltop streams receive inputs from precipitation, overland flow, and local hilltop aquifers. These local aquifers are formed by shallow groundwater perched on low-permeability rock layers above coal seams. Lateral discharge into hillside streams and springs, often at coal outcrops (see Figure 5), is the primary direction of flow from such aquifers, which can have residence times as brief as a week (Hawkins et al., 1996). Vertical discharge through rock fractures (secondary porosity) and intact layers (primary porosity) also occurs but typically at much slower flow rates. Vertical discharge occurs through deep zones of unweathered rock and has long residence times, while lateral discharge circulates on near-surface zones of weathered rock with shorter residence times. Both types of flow connect headwaters and local aquifers to regional rivers and groundwater. Hilltop stream channels and aquifers slow runoff into valleys, reducing erosion and contributing to flood control (Callaghan et al., 2000).

Here we summarize the available data on headwater ecosystem loss and burial (see Section 3.2); and potential impacts to headwater biota (see Section 3.3) and headwater ecosystem function (see Section 3.4 and Figure 9).

3.2. ESTIMATING THE EXTENT OF HEADWATER ECOSYSTEM LOSS

The OSMRE inventoried valley fills in the central Appalachian coalfields to estimate the number of headwater stream miles lost to mountaintop mining and valley fills, based on permit data and 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 mi) 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 those 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 mi) of streams were approved to be lost due to mountaintop removal, 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 loss or burial during this 10-year period. More current statistics were unavailable at the time this report was written, but both mine footprints and stream losses were projected to double over 2002 levels by 2012 (U.S. EPA, 2002).

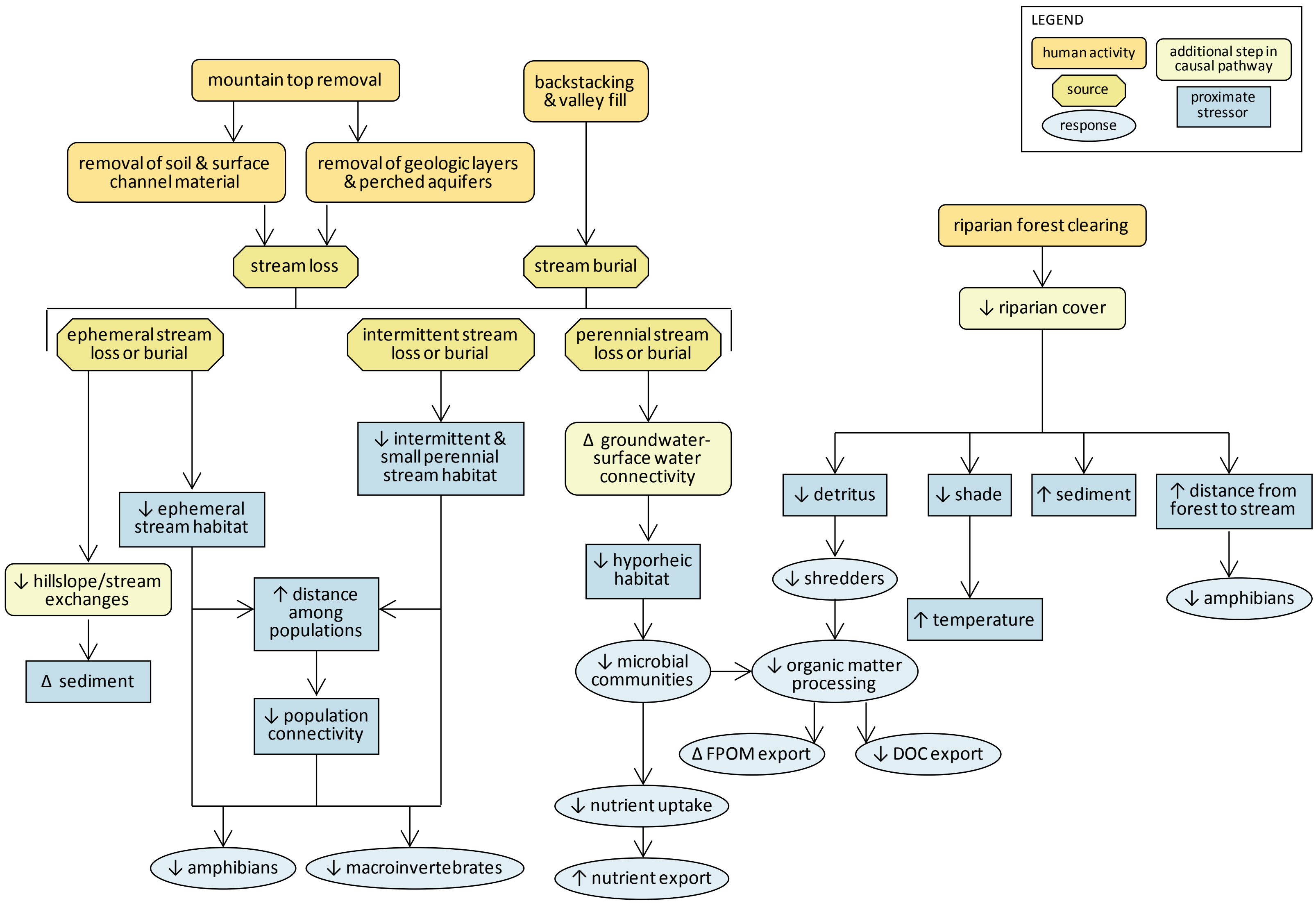

Figure 9. Observed and expected effects of stream loss and burial and riparian forest clearing on aquatic ecosystems. See Section 3 for more discussion and evidence.

Headwater streamflows are tightly coupled with catchment and hillslope processes (Gomi et al., 2002). Headwater catchment burial estimated as the watershed area above the toe of valley fills on permits approved from 1985 through 2001 is another metric for assessing headwater ecosystem loss (see Table 1).

Table 1. Watershed areas above the toe of valley fills in permits approved between 1985 to 2001

State	Average watershed area above valley fill toe km^2 (acres)		Largest watershed area above valley fill toe km^2 (acres)		Total watershed area approved for valley fill km^2 (acres)	
KY	0.23	(56.30)	15.28	(3,777)	1,138.57	(281,347)
TN	0.22	(54.85)	1.17	(288)	12.21	(3,017)
VA	0.33	(81.05)	5.01	(1,238)	172.5	(42,629)
WV	0.39	(97.28)	6.59	(1,628)	451.14	(111,479)
Total					1,774.42	(438,472)

Source: Chapter III of EPA (2003, 2005).

At the time of this report, data to quantify the area impacted by valley fill permits approved prior to 1985 or since 2001, or predict cumulative losses from planned MTM-VF activities, are not available in the MTM-VF PEIS or peer-reviewed literature. Data to quantify headwater stream loss by flow duration class (e.g., ephemeral, intermittent, and perennial) are not available; however, in a study of 36 headwater streams in southern West Virginia for which valley fill permits were pending or approved, Paybins (2003) estimated that the median watershed area for intermittent flows was 0.06 km^2 (14.5 acres) and the median watershed size for perennial flows was 0.17 km^2 (40.8 acres). The same study cites digital geographic information system (GIS) data of valley fills estimated from permit maps by the West Virginia Department of Environmental Protection (WVDEP) showing that the median size of permitted fills in southern West Virginia is 0.05 km^2 (12.0 acres, comparable to the median intermittent stream drainage), and the maximum size of permitted fills is 1.94 km^2 (480 acres). Statewide averages of valley fill area reported in the MTM-VF PEIS (see Table 1) range from 0.22−0.39 km^2 (54.36−97.28 acres). Estimates of valley fill area from both reports suggest that headwaters in all three duration classes are being permitted for burial by valley fills. Valley fill footprints estimated from permit maps may be smaller or larger than the actual fill areas. Some areas approved for fill may not be used; in other cases, areas from permit maps underestimate the actual area of fill (see Figure 10).

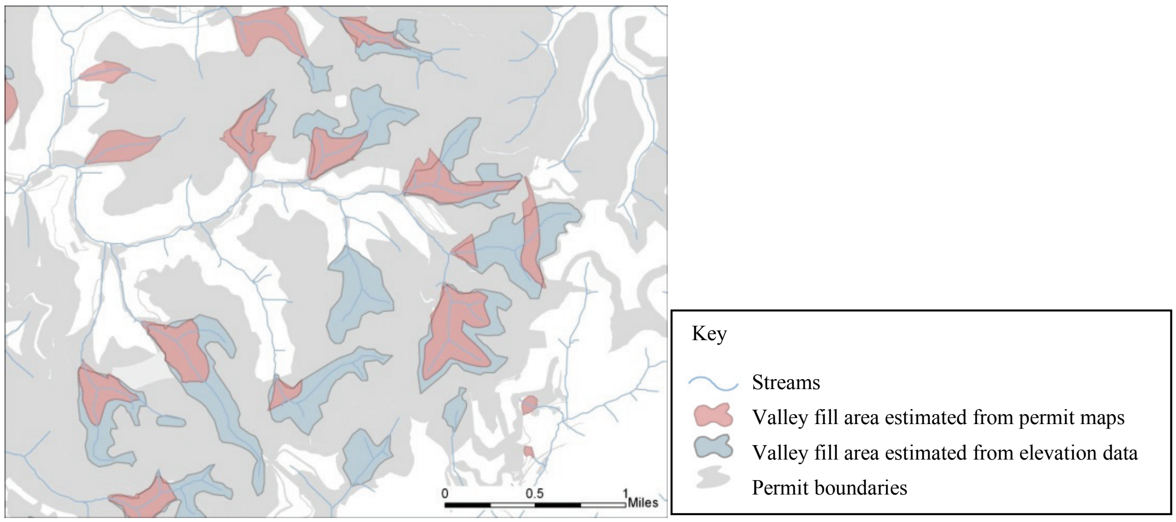

Figure 10. Map showing loss of headwater streams to MTM-VF. This diagram depicts the loss of stream miles and channel complexity that can result from extensive mountaintop mining and valley filling. Solid blue lines inside valley fill areas represent buried streams. Note that some headwaters above filled areas are disconnected from the rest of the stream network.

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

Estimating headwater stream loss in terms of area or miles of stream impacted in watersheds above a size threshold is a useful beginning, but it does not address the full extent of affected headwaters, or loss of other aquatic ecosystems. For example, the current estimate does not include unmapped streams, springs, seeps, and wet areas that may occur in watersheds less than 0.12 km^2 (30 acres) in size, or headwaters disconnected from the stream network by valley fills (see Figure 10). Small stream channels often are not designated on United States Geological Survey topographic maps (Hansen, 2001) and difficult to detect on aerial photographs; thus, accurate inventories of them are difficult and surveys frequently underestimate their true extent (Fritz et al., 2006). Similarly, estimates based on stream miles or catchment area do not include impacts due to the loss of headwater wetlands and forested vernal pools, which provide refuge and habitat for breeding, hunting, foraging by amphibians, reptiles, and aquatic or semiaquatic invertebrates. 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 riparian habitat in the study area. Approximately 42%

of these projected losses occur in headwater (first- and second-order) streams (U.S. EPA, 2002). Data for forested wetlands, and for riparian habitats lost in other MTM-VF states, were not available at the time of this report.

Further, current estimates of impacts do not account for potential long term effects of landscape-scale changes in land cover associated with mountaintop removal mining. Effects of spatially extensive changes in land-cover on regional biodiversity can persist for decades. For example, Harding et al., (1998) found that past land-use activity (primarily deforestation and agriculture) resulted in long term modifications to and reductions in stream fish and invertebrate diversity, despite preservation of forest patches and reforestation of stream riparian zones.

3.3. 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 period, which ended 10,000 years ago (McKeown et al., 1984; Soltis et al., 2006; Zeisset and Beebee, 2008; Potter et al., 2010). The New and Kanawha Rivers in West Virginia were the headwaters of the Teays paleodrainage, a major preglacial route of fish dispersal from the Appalachian Mountains to the Mississippi River, and they are home to many endemic (regionally unique) species today (Hocutt et al., 1978; Stauffer and Ferrerri, 2002). New stream species or potential species are still being discovered in central Appalachia, which includes areas of notable biodiversity identified by NatureServe (see Figure 11). For example, Berendzen et al. (2008) found that the roseyface shiner (*Notropis rubellus*), thought to be a single, widespread species, includes at least four species in the Central Highlands and Lowlands with an endemic genetic lineage above Kanawha Falls. Kozak et al. (2006) found high levels of genetic diversity in the common two-lined salamander (*Eurycea bislineata* complex), including an endemic lineage in southern Virginia and West Virginia. Nearly 10% of global salamander species diversity is found within streams of the southern Appalachian Mountains (Green and Pauley, 1987), which are near the southern extent of the Teays River paleodrainage. A phylogeographic study of the North American giant salamander or hellbender (*Cryptobranchus alleganiensis*) found evidence of close genetic relationships among populations living in the central Appalachian, southern Appalachian, and Ozark Mountains, which were connected via preglacial river systems (Routman et al., 1994). The biodiversity of this region is a valuable natural resource in economic, cultural, aesthetic, scientific, and educational terms (Hughes and Noss, 1992; Cairns and Lackey, 1992; Dudgeon et al., 2006; Meyer et al., 2007).

We assume that most of the organisms inhabiting a headwater stream and riparian area are eliminated when that headwater basin is buried or blasted during the mining process. It is possible that some microorganisms persist in or colonize buried stream channels, but we found no studies of these systems. Surveys conducted as part of the PEIS, studies of stream biota found in unmined Appalachian streams, and relevant studies of headwaters in other temperate regions provide background on potential biodiversity impacts due to MTM-VF. This information is discussed below.

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 microorganisms (Southerland, 1986; Vinson and Hawkins, 1998; Meyer et al., 2007). Communities in small, permanent or intermittent streams differ from those found in larger streams and rivers (Vannote et al., 1980; Morse et al., 1993, 1997; Hakala and Hartman, 2004; Stauffer and Fererri, 2002). Flow permanence and duration are likely to influence aquatic community structure, because the relative abundance (number of individuals) of species with lower resistance or resilience to drying is expected to decrease as surface water flows become more intermittent (Fritz and Dodds, 2004; Arscott et al., 2010). Many headwater stream taxa are adapted to variable flows, and even ephemeral and intermittent streams can support diverse and abundant invertebrate assemblages (Feminella, 1996; Williams, 1996; Kirchner et al., 2003). Kirchner et al. (2003) sampled 36 intermittent and perennial headwater 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 the genera were found in both intermittent and perennial stream types. Similarly, Collins et al. (2007) found that subsurface invertebrate community composition was comparable in intermittent and perennial stream reaches of a stream having surface flows over only 30% of its length in summer.

In addition to the differences in taxonomic structure of invertebrate assemblages in headwater streams, the functional role of aquatic invertebrates also differs from larger streams and rivers (Vannote et al., 1980) because of the closer contact between stream and forested terrestrial ecosystems. Particularly in densely forested Appalachian catchments, large amounts of leaf litter fall into small streams in autumn. This leaf litter supplies much of the energy for the detrital food webs in these streams (Wallace et al., 1997; Meyer and Wallace, 2001; Cross et al. 2006; Wipfli et al., 2007). Many invertebrates in small streams consume whole, decomposing leaves (i.e., shredders) or organic particles created when these leaves are broken apart (i.e., collectors). In turn, predators, including other invertebrates, salamanders, and fish, feed on these invertebrates.

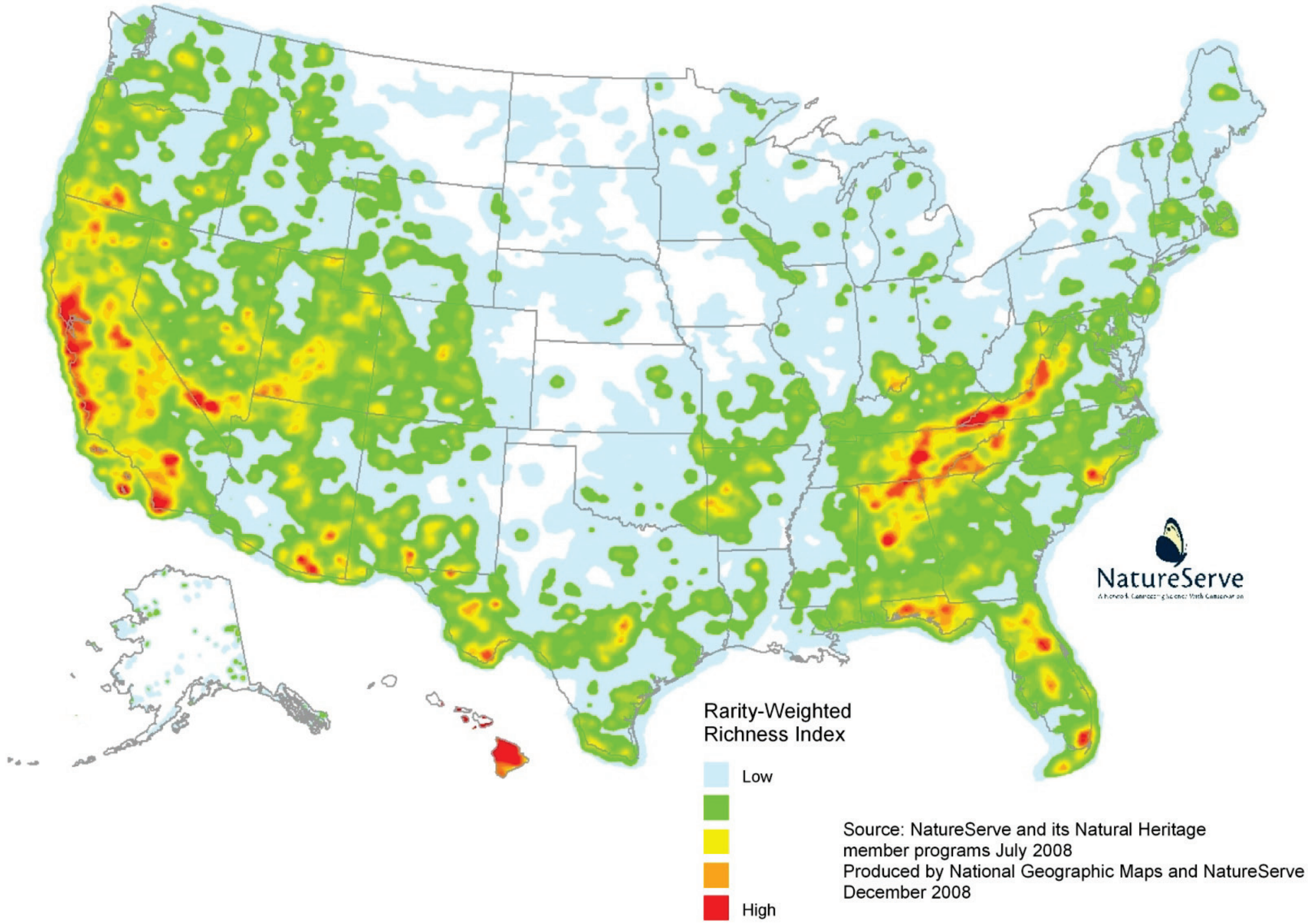

Figure 11. Hot spots of rarity-weighted species richness in the United States. The central Appalachian Mountains—including the central Appalachian coalfields—have been identified as one of the most significant hot spots for biological diversity in the United States.

Source: NatureServe and its Natural Heritage member programs, July 2008 (National Geographic Maps and NatureServe, 2008). Used with permission.

Headwater streams support diverse algal and fungal communities. In studies of two Appalachian headwater streams, more than 30 species of diatoms and more than 40 species of fungi were recorded (Gulis and Suberkropp, 2004; Greenwood and Rosemond, 2005). Many algal and fungal species' ranges extend from the Mid-Atlantic Highlands to the southern extent of the Appalachian Mountains (e.g., Ponader and Potapova, 2007), and similarly high levels of diversity are expected in central Appalachia (Pan et al., 2000). Diatoms (silica skeleton-producing algae) and fungi are important food sources for fish and aquatic insects. 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 makes energy and nutrients in difficult-to-digest vegetation accessible to fish and invertebrates (Gulis et al., 2006).

Appalachian headwater streams also support diverse and abundant assemblages of amphibians. Salamanders are the most common vertebrates in headwaters and may often be the major predator of the aquatic invertebrates (Davic and Welsh, 2004). Many stream salamanders require ephemeral and intermittent streams in forested habitats to maintain viable populations (Petranka, 1998; Davic and Welsh, 2004). 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 very small permanent or intermittent streams because these reaches are too small to support predatory fish (Petranka, 1983; Davic and Welsh, 2004). For example, in a radio-telemetry study of black-bellied salamanders (*Desmognathus quadramaculatus*) in one spring-fed stream, Peterman et al. (2008) estimated the population density to be 11,294 salamanders per hectare (2.47 acres), or 99.30 kg per hectare of biomass. 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).

Although some of these species occur in larger streams downstream from MTM-VF, Appendix F of the PEIS lists 22 fish, 1 salamander, 1 bird, 38 mussels, 7 snails, and 6 aquatic invertebrates that are considered threatened, endangered, or species of special concern in the central Appalachians that are associated with streams (FWS, 2003). Some 31 plants and 21 terrestrial and 50 cave-dwelling invertebrates are also listed.

Loss or burial of headwater streams and associated riparian and subterranean ecosystems can result in fragmentation of remaining habitats by increasing geographical distance among populations. Subdivided populations are smaller in size, and thus more susceptible to loss of genetic diversity and to adverse effects of environmental change, placing them at higher risk of local extinction (Gilpin and Soulé, 1986; Frankham, 2005). The effects of fragmentation may be increased by the dendritic structure of stream networks (Fagan, 2002). Loss of aquatic organisms from MTM-VF-impacted streams has been documented (e.g., Pond et al., 2008; Pond, 2010), but, to our knowledge, effects of habitat fragmentation on surviving populations have not been studied in the central Appalachians. Relevant studies from other areas indicate potential for detrimental effects on species that disperse frequently among headwater streams—including salamanders (Grant et al., 2010) and insects (reviewed by Hughes et al., 2009); and species that disperse longitudinally through stream networks, especially fish (Fagan, 2002; Letcher et al., 2007).

For species capable of overland dispersal, network configuration and terrestrial land use in headwater catchments can increase or decrease connectivity or isolation of stream populations. Alexander et al. (2011) found that tree cover in first-order watersheds was the best predictor of regional genetic diversity in the common mayfly *Ephemerella invaria*, which is closely related to ephemerellid species in central Appalachia (Alexander et al., 2009). 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 (Petranka et al., 1993; Maggard and Kirk, 1998). Grant et al. (2009) found higher occupancy by salamanders in less-developed catchments of headwaters connected to other headwaters (i.e., more highly branched headwater networks).

3.4. LOSS OR ALTERATION OF HEADWATER ECOSYSTEM FUNCTIONS

As with the loss of biota, we assume that most ecosystem functions performed by a high-gradient, forested Appalachian headwater stream are lost when it is buried or removed. Some functions, such as water conveyance and export of dissolved solids, might continue under fills in a quantitatively or qualitatively altered state. At the time of this report, no ecosystem studies of buried streams were found in the published literature. Evidence of stream function in channels constructed on valley fills is reviewed in Section 7.

Data to quantify regional loss of stream function due to mountaintop removal and valley fill mining are not available at the time of this report. Here, we briefly review ecologically important functions that were likely to have been operating in central Appalachian headwater streams lost to blasting or burial.

3.4.1. Transformation and Removal of Nutrients and Contaminants

Due to the small size of headwater streams, their contributions to ecosystem function at the watershed scale are often overlooked. However, the individual and cumulative effects of headwater streams can be substantial (Meyer and Wallace, 2001; Benda et al., 2004; Freeman et al., 2007; Meyer et al., 2007). Nutrients are taken up and transformed more rapidly in headwaters, where waters slowed by woody debris 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 microbes perform in the substrates and hyporheic zones of natural stream channels and riparian areas (Payne, 1981), removes N from the stream in the form of N gases 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 microbial communities in the riparian-hyporheic zone (Pusch et al., 1998; Mayer et al., 2007).

In addition to reducing excess nutrients, natural headwaters can remove metal contaminants including 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 function and the export of toxicants act in combination to increase risks to water quality below MTM-VF.

3.4.2. Storage and Export of Woody Debris

In their natural state, forested headwaters typically transport little sediment or 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 provide habitat and 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. Woody debris could be of particular importance in this, because both phosphorus and ammonium can travel further downstream before being taken up by benthic organisms when wood is removed from a headwater stream (Webster et al., 2000).

3.4.3. Organic Matter Processing

Forested headwaters also receive and process large volumes of organic matter from upland and riparian vegetation (Wipfli et al., 2007). 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,

[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 flora and fauna (Boulton et al., 1998).

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; Hall et al., 2000; Wipfli et al., 2007). Macroinvertebrates and detritus from headwater streams supports the biomass of animals, plants, and fungi found in downstream segments (Wipfli and Gregovich, 2002; Brittain and Eikeland, 1988). The organic matter transported from headwaters to downstream segments is largely in the form of fine particulate organic matter (FPOM, 0.63 μm to 1.0 mm diameter) or dissolved (DOM) organic matter (Wipfli et al., 2007). Kaplan et al. (2008) found that only 8.6% of dissolved organic carbon (DOC) was labile and was taken up within a stream reach, while the remaining bioavailable DOC was semi-labile and is usually transported out of the reach to larger streams. The loss of trophic subsidies from headwater streams may lead to lower secondary productivity in downstream habitats.

3.4.4. Habitat

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). These areas could 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 Kodric-Brown, 1977). Headwaters also serve as nurseries and spawning grounds for amphibians and fish, including the brook trout (*Salvelinus fontinalis)*, creek chub (*Semotilus atromaculatus*), blackside dace (*Phoxinus cumberlandensis*), southern redbelly dace (*Phoxinus erythrogaster*), arrow darter (*Etheostoma sagitta*), and orangethroat darter (*Etheostoma spectabile* [Meyer et al., 2007]). Today, most brook trout populations in West Virginia occur outside of the coal mining region, but there is overlap in their historic range. In a study of one West Virginia watershed, Petty et al. (2005) estimated that >80% of all brook trout spawning occurred in small streams (watersheds <3 km^2), including headwaters draining areas less than 0.25 km^2.

4. IMPACTS ON WATER QUALITY

In this section, we report the results of a number of studies that have assessed the changes in the physicochemical attributes of streams downstream of MTM-VF. Although much of this information might also apply to constructed channels and other water-containing structures on the valley fills and the mined site, there are few data in the PEIS or the peer-reviewed literature on these constructed systems. The physicochemical attributes we review below include alteration of streamflow, water chemistry, sedimentation of stream substrates, 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 STREAMFLOW

Four factors can affect streamflow below valley fills. First, trees and other vegetation are removed from both the mined area and the area of the valley fill, the reclaimed surface might be planted with grasses, and, if planted, trees are generally slow to regrow on the reclaimed mined area and valley fill (see also Section 7). This reduces evapotranspiration rates from the watershed because transpiration is a function of the type and abundance of 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 might 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 a headwater stream is lost (see Section 3), attributes that influence surface flow (e.g., woody debris, surface water/ground water connections) are also lost (see Figure 12). Valley fills can 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 flow in the mined watershed was greater than that in the unmined watersheds during summer, autumn and early winter (July to January), when soil and aquifer moisture levels were reduced (see Figure 5, 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 unmined watersheds during periods

[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.

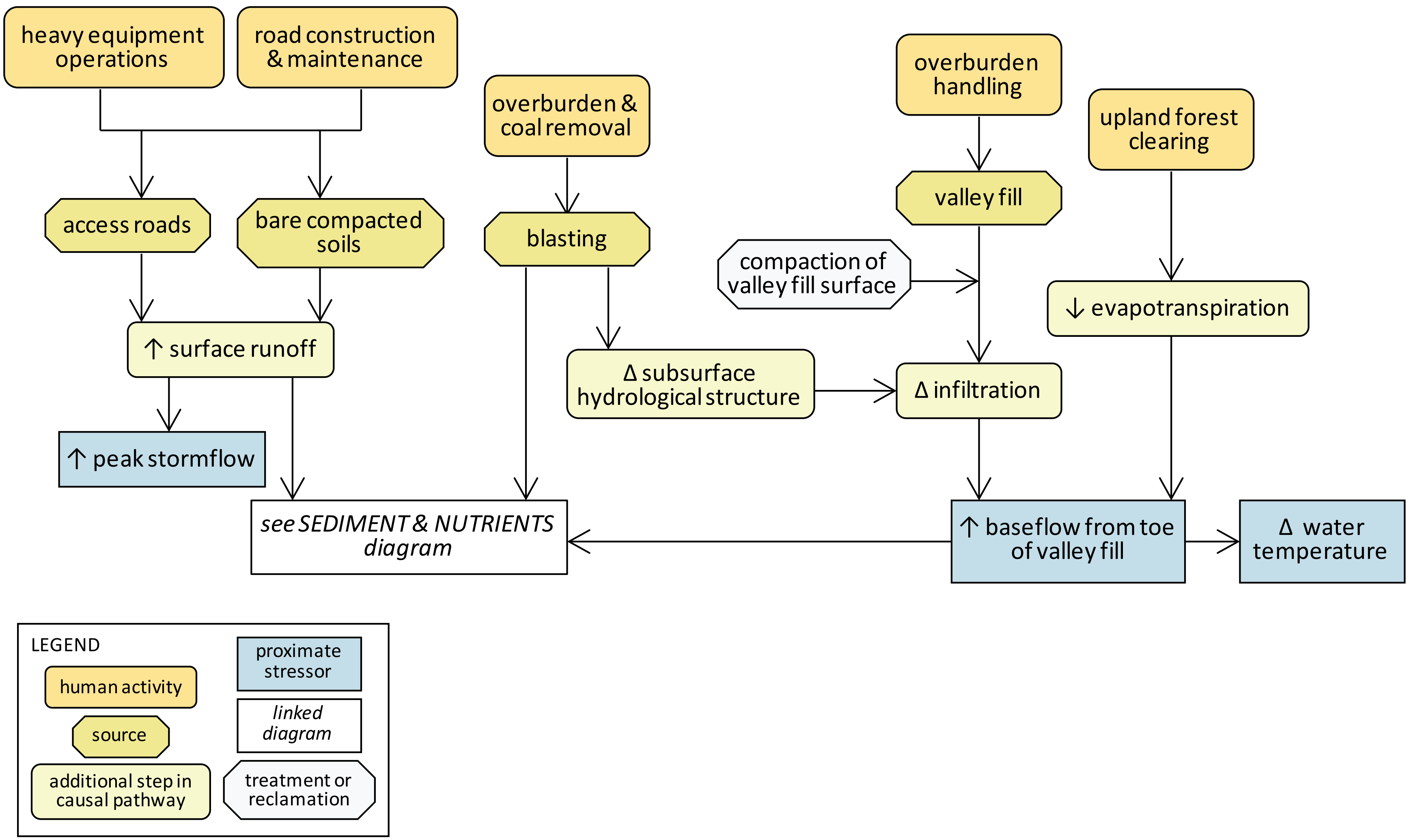

Figure 12. Observed and expected effects of MTM-VF on streamflow characteristics. See Section 4.1 for additional details and evidence.

of low streamflow (Wiley et al., 2001). Both Green et al. (2000) and Armstead et al. (2004) observed that the streams below valley fills continued to have surface flows during the summer and fall of a year when a drought occurred, but several of their unmined sites did not 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 might 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 7.6 to 15 cm 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). Peak discharges at the third site downstream of a single, large valley fill without active mining had an estimated flood recurrence interval of <2 years, much less severe than the other two mined sites. The differences might be due to unaccounted for differences in rainfall among the watersheds or differences in mine and valley fill attributes. Thunderstorms can cause locally variable rainfall, particularly in mountainous terrains (Barros and Lettenmaier, 1994; Roe, 2005).

Some current state regulations generally require that MTM-VFs be designed to reduce such increases in downstream flooding (WVDEP, 2009b). Recent research by Taylor et al. (2009a, b) suggests that alternate spoil placement methods that reduce compaction of the mining spoil might increase infiltration of rainfall and ameliorate the increases in peak discharges.

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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 are 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 CHEMICAL TRANSPORT

4.2.1. 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 in a reaction catalyzed by autotrophic bacteria to form the strong acids characteristic of acid mine drainage (Stumm and Morgan, 1996):

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

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 can 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 2−5). 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 et al., 1997; Skousen et al., 1997). Other methods that can moderate pH include physically isolating the pyritic materials within the mine or valley fill (Skousen et al., 2000; Hawkins, 2004) and treatment within the sediment retention pond (Hartman et al., 2005; see Table 4 and Figure 13).

Iron forms relatively insoluble compounds, such as $Fe(OH)_3$, under more alkaline conditions and might not be found in elevated concentrations in the effluent waters below valley fills (Bryant et al., 2002; see Table 2). However in some conditions, such as during higher flows, Fe can remain elevated (Hartman et al., 2005; see Table 4).

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 et al., 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 2). Exceptions to this are Mn and nickel (Ni), which can be elevated in the effluent waters below valley fills (Bryant et al., 2002; Hartman et al., 2005; see Tables 2 and 4). Mn can occur in association with siderite ($FeCO_3$) in shales

Table 2. Water quality variables in unmined streams versus streams below valley fills. Variables are ordered by ratio of their median concentration in unmined streams to those with valley fills (Filled). Units are mg/L, unless indicated otherwise.

Variable	Unmined			Filled			Detection limit
	Median	Mean	Range	Median	Mean	Range	
SO_4^{2-}*	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^-*	---	25.5	7.44–42.7	---	223	13.1–612	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

Table 2. Water quality variables in unmined streams versus streams below valley fills (continued)

Variable	Unmined			Filled			Detection limit
	Median	Mean	Range	Median	Mean	Range	
Cd, total	<0.0005	---	---	<0.0005	---	---	0.001
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

Sources: Bryant et al. (2002) and 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 $p = 0.05$. Median concentrations (Bryant et al., 2002) are from 9 unmined sites and 21 filled sites, each sampled about six times from August 2000 to February 2001. Means and ranges (Pond et al. 2008) are from sites having 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. In Pond et al. (2008), 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^- by multiplying by 1.22. Concentrations below detection are shown as <½ the detection limit. A "---" under median, mean, or range indicates that this variable was not reported. 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]). 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.

Table 3. Water quality parameters for unmined or reference streams or streams downstream from mined, filled, or filled and residential watersheds in West Virginia

	Green et al. (2000)				Merricks et al. (2007)		Hartman et al. (2005)	
Variable	**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 4. 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
Mn*	0.019	0.0016–0.046	0.062	0.0020–0.17
Fe*	0.016	0.0014–0.030	0.047	<0.00050–0.082
Zn	0.0027	0.0014–0.0047	0.0028	0.00090–0.0086
Al	0.012	0.0090–0.019	0.019	0.00090–0.064

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 5. Range of dissolved oxygen, pH, and conductivity values 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

Source: Howard et al. (2001).

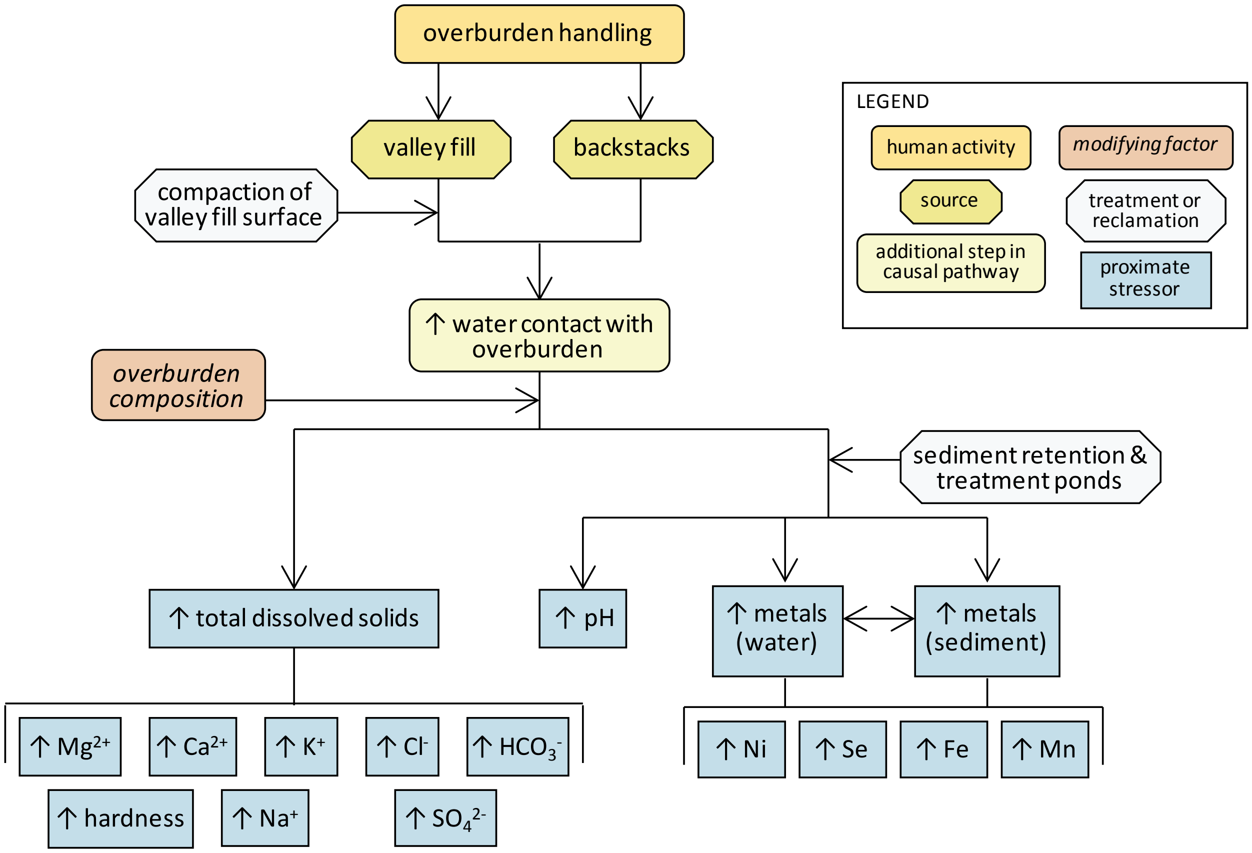

Figure 13. Observed and expected effects of MTM-VF on total dissolved solids, metals, and pH. Water quality variables shown were significantly different from reference sites in at least one of the reviewed studies. See Section 4.2 for additional details and evidence.

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 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 and dominate the mixture of ions in these waters in relative concentrations (Bryant et al., 2002; Hartman et al., 2005; see Tables 2 and 4). In addition, other water-soluble compounds within coal or overburden can be solubilized by the above reactions or just by the increased exposure to water in the fragmented overburden (Yudovich and Ketris, 2006a; Vesper et al., 2008). These ions, including K^+, Na^+, and Cl^-, and Se, occur at elevated concentrations in the effluent waters (see Tables 2 and 4), but at concentrations at least one order of magnitude less than those observed for SO_4^{2-}, Ca^{2+} , Mg^{2+}, and HCO_3^-.

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 et al., 2007; Merricks et al., 2007; Pond et al., 2008; see Tables 2–5). In these studies, measured conductivity at sites downstream from valley fills ranged from 159−2,720 μS/cm and at unmined reference sites from 30−260 μS/cm. Hardness is another measure of these dissolved ions—particularly the divalent ones like Ca^{2+} and Mg^{2+}. These aggregate measures of total ions are coarser than the individual ion concentrations but are relatively simple to measure. TDS theoretically would be simply the sum of the dissolved ion concentrations (mg/L), but, in practice, it also includes some particulates that pass through the filter used to separate dissolved solids from suspended solids (APHA et al., 1998).

In terms of individual dissolved ion concentrations, conductivity is a product of the molar concentration of each ion (mmol/L), the absolute value of its charge (meq/mmol), and an ion-specific, equivalent conductance (λ°) that is 80.0, 59.5, 53.1 and 44.5 μS/cm^2/meq for the dominant ions, SO_4^{2-}, Ca^{2+}, Mg^{2+}, and HCO_3^-, respectively, at 25.0°C (APHA et al., 1998; Talbot et al., 1990; Pawlowicz, 2008). The equivalent conductances for the less elevated ions, K^+, Na^+, and Cl^-, are 73.5, 50.5,76.4 μS/cm^2/meq, respectively, at 25.0°C (APHA et al., 1998). For example

$$\text{Conductivity } [SO4^{2-}] = \text{Molar Conc. (mmol/L)} \times 2 \text{ meq/mmol} \times 80\ \mu\text{S/cm}^2\text{/meq.} \quad \text{(Eq. 4)}$$

Considering the greater concentrations of the four dominant ions, SO_4^{2-}, Ca^{2+}, Mg^{2+}, and HCO_3^-, and the greater charge of three of these ions, it follows that these ions also dominate the ion mixture in their contribution to the conductivity.

Most studies have not assessed the seasonal variability of water chemistry at these sites, but Green et al. (2000) present 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 6). 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 sites in filled and filled/residential watersheds were an order of magnitude (10 times) greater than at reference sites in unmined watersheds (see Table 6). Pond et al. (2008) observed conductivities up to 2,540 μS/cm in streams from mined watersheds.

Table 6. 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).

Moreover, only one study has assessed the downstream extent of elevated ion concentrations related to these sites but only within a single stream drainage. Johnson et al. (2010) measured conductivity at intervals of 100 to 500 m along the main stem of Buckthorn Creek (Breathitt, Perry, and Knott Counties, Kentucky) downstream from a valley fill and along several mined and unmined tributaries. Maximum conductivities along Buckthorn Creek were 3,190 µS/cm in spring (May 2006) and 11,810 µS/cm in summer (September 2005), and measured conductivities gradually decreased in a downstream direction—except downstream from the mined tributaries. Downstream of mined tributaries, conductivities increased, while downstream of unmined tributaries, conductivities decreased. The increase or decrease was related to the watershed size of the tributary, which was used as a surrogate measure of the relative discharge of the tributary. This suggests that most of the decrease in conductivities was the result of dilution by low conductivity, unimpacted waters. Downstream from the first valley fill along the mainstem of Buckthorn Creek (i.e., a distance of 20 km), conductivities never decreased below 400 µS/cm in spring and 2,000 µS/cm in summer.

Kirk and Maggard (2004) sampled a third-order stream, Trough Creek, at two stations: one was upstream of the stream's confluence with two smaller tributaries where mountaintop mines and valley fills were developed, and the second was downstream of both confluences. Sampling was conducted in spring (i.e., April) and autumn (i.e., October) from October 1995 (before mining began in February 1996) until April 2003. The downstream sites exhibited variation—particularly for SO_4^{2-}, TDS, and conductivity—between the two seasons that appears to be related to dilution associated with seasonal variation in discharge of Trough Creek (Kirk and Maggard, 2004). Conductivity in spring increased from 64 µS/cm in 1996 to over

400 µS/cm in 2002 and 2003, while in autumn, conductivity increased from 242 µS/cm in 1996 to over 1,600 µS/cm in 2002 and 2003 (Kirk and Maggard, 2004).

The relative ion composition of the effluent is expected to be fairly consistent among valley fills in the region. Although the names of coals vary by state, mountaintop mines in the central Appalachians primarily mine a similar series of coals that occur above the level of the permanent draining streams. Specific coals include the No. 6 Block (called Richardson and Skyline in Kentucky), Stockton (called Broas in Kentucky), Coalburg (called Peach Orchard in Kentucky), and in some cases the Winifrede (called Hazard and Haddix in Kentucky), Chilton (called Cropland and Taylor in Kentucky), and Fire Clay (Neuzil, 2001; Tewalt et al., 2001). These coals occur in the lower Allegheny (called Charleston in West Virginia and upper Breathitt in Kentucky) and Pottsville (called middle to upper Kanawha in West Virginia and middle to lower Breathitt in Kentucky) formations, both of which formed during the middle Pennsylvanian period. The intervening noncoal formations, which form the overburden removed by mountaintop mining and placed into the valley fills, are siltstones, shales, and sandstones with a few limestones, mostly of marine origin. Because of the mixing of overburden formations within the valley fill and the geochemical reactions that create the ion mixture, the relative ion composition of the effluent is unlikely to differ significantly among valley fills in the region. However, the maximum conductivities might differ substantially (Green et al., 2000; Howard et al., 2001; Bryant et al., 2002; Bodkin et al., 2007; Merricks et al., 2007; Pond et al., 2008), in part as a function of the amount of dilution by unaffected, lower conductivity waters (Johnson et al., 2010).

Se is enriched in coal relative to other rocks (Coleman et al., 1993). In coals mined by MTM, average Se concentrations range from 3.9 to 7.1 µg/kg in West Virginia (Neuzil et al., 2005) and 3.8 to 6.6 µg/kg in Kentucky (Eble and Hower, 1997). It appears to be mainly associated with the organic fraction of the coal, where it substitutes for organic sulfur. It can also be associated with pyrite or with other accessory minerals, clausthalite (lead selenide) (Coleman et al., 1993; Finkelman, 1994; Hower and Robertson, 2003, Yudovich and Ketris, 2006b). Reflecting these different modes of occurrence, the correlations between Se and the organic content of the coal as measured by loss on ignition or total sulfur (e.g., reflecting the pyrite content of the coal) are variable among regions (Coleman et al., 1993), but Neuzel et al. (2007) found significant correlations between Se and the organic content of coal-bearing strata (i.e., noncoal formations adjacent to coal measures that form the overburden) from the central Appalachians, but not between Se and total sulfur. Neuzel et al. (2007) did not find a correlation between the Se concentration in these rocks and the Se concentration in leachates.

4.3. OTHER WATER QUALITY VARIABLES

Other water quality variables investigated include water temperature, nutrients, particularly nitrogen and phosphorus, and dissolved oxygen (see Figure 14).

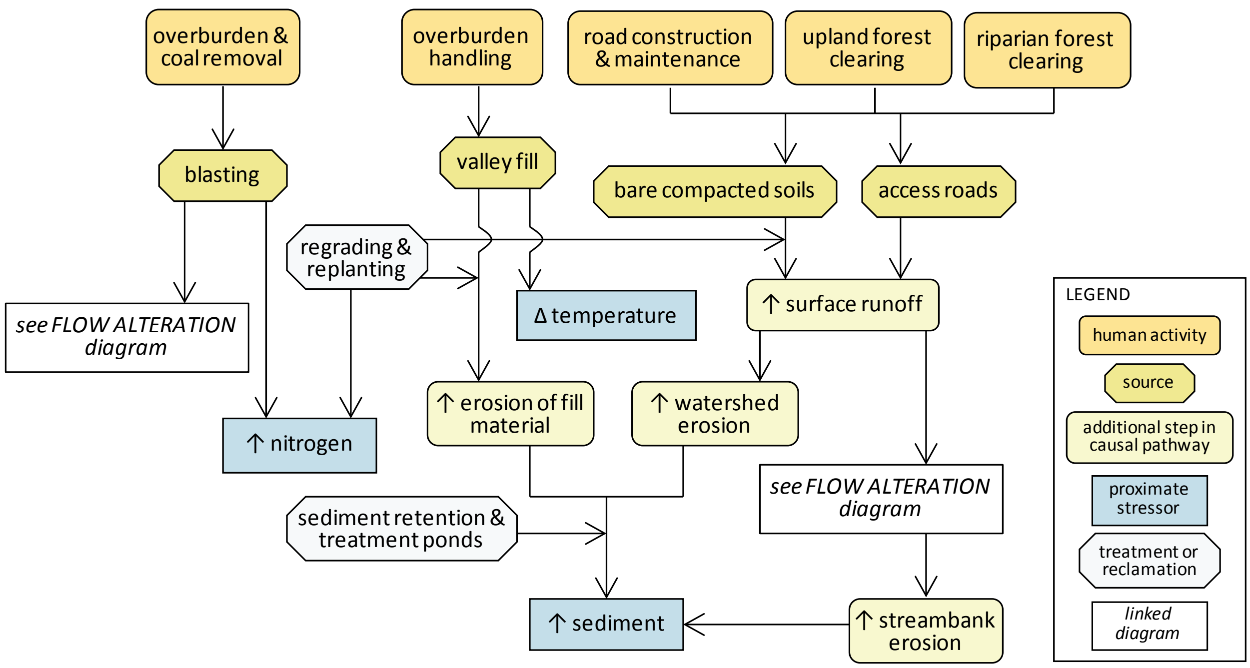

Figure 14. Observed and expected effects of MTM-VF on sediments, nutrients, and temperature. See Sections 4.3 and 4.4 for additional details and evidence.

4.3.1. 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.

4.3.2. 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).

4.3.3. Dissolved Oxygen

In the studies that have measured dissolved oxygen, 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 3 and 5; 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 SEDIMENTATION

Sediment retention ponds built downstream of valley fills are intended to capture sand and finer-sized particles that are produced by the fragmentation of the overburden and washed downstream from the toe of the valley fill (U.S. EPA, 1979; see Figure 14). 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 valley fills and any sediment retention ponds (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 smaller in filled or filled/residential streams downstream from sediment retention ponds compared to unmined streams, and the mean percentage of sand and fines was

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[6]The bankfull channel is the entire channel, which is submerged at bankfull discharge—the point just before the streamflow 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.

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 7).

Table 7. Substrate measures in streams located in different land use classes

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 and fines)	16.9 (9.9)	20.7 (12.9)	29.7 (24.1)	8.0 (9.2)

Source: Green et al. (2000).

Hartman et al. (2005) did not find any clear pattern of fine sediments in a study that compared pairs of mined and unmined sites using samples taken in December with a scoop sample separated with modified Wentworth sieves (McMahon et al., 1996; see Table 8). In two cases, the proportions of sand and fines were similar; in the third case, it was greater in the filled site; and in the fourth 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. In this study, the filled sites were upstream from the sediment retention ponds (Hartman et al., 2004).

Table 8. 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).

Much of the fine sediment, though, might come from the streambanks rather than the mined area or the valley fill. Using stable isotopic signatures of carbon (i.e., $\delta^{13}C$) and nitrogen (i.e., $\delta^{15}N$), Fox (2009) attempted to identify the sources of fine sediments in streams in four forested catchments. Of the catchments, one had an active mountaintop mine, one had an inactive mountaintop mine, one had some pre-SMCRA surface mining, and one had no surface mining. In the mined watersheds, there were two potential sources of fine sediments: streambank erosion and surface erosion from the mined area and valley fill. Fox (2009) concluded that about 50% of the fine sediments in the inactive mined watershed and 40% in the actively mined watershed were from streambank erosion. The streambank erosion could occur because of the alteration in stream baseflows and peak flows caused by mining and creation of the valley fills (see Section 4.1).

4.5. CHANGES IN SEDIMENT CHEMISTRY

Data on sediment chemistry in larger streams downstream of valley fills are limited to a study by Merricks et al. (2007), who measured metals and arsenic. 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).

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).

Sediment concentrations of metals and arsenic 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.

4.6. CUMULATIVE IMPACTS

In terms of downstream water chemistry, the primary cumulative impact of MTM-VF and other mining methods for coal 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 that alter these ion concentrations in the waters. Any changes in SO_4^{2-} concentrations are the outcome of mixing of waters with differing SO_4^{2-} concentrations (Cooper et al., 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. MTM-VF appears to be these sources because other land disturbances, such as residential development, are not origins of elevated SO_4^{2-} and conductivity.

Conversely, while total Fe, total Mn, and total Al in many larger streams within mined basins exceeded regional background concentrations of 129, 81, 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 might reflect the increase in pH associated with the increased number of valley fills, which are the sources of alkaline waters within the Kanawha basin.

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 first-order, headwater streams contributed 70% of the mean annual water volume in second-order streams and 55% of the volume in higher-order rivers. For nitrogen, a nutrient that is not as conservative as the ions associated with

MTM-VF, these first-order streams contributed 65% of the flux in second-order streams and 40% of the flux in higher-order rivers (Alexander et al., 2007).

Johnson et al. (2010), working in a single stream drainage with some tributaries directly affected by valley fills and others that were not, measured conductivities upstream and downstream of the tributary confluences. They found that the cumulative level of conductivity values could be predicted using a simple function that combined tributary concentrations with watershed area. Watershed area was a surrogate for volume of discharge from the tributary.

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). 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 (Paybins et al., 2000). 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 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. AQUATIC 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 measure the effect of 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 significant limitations for evaluating MTM-VF effects: neither *C. dubia* nor *P. 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). On the other hand, toxicity tests might overestimate effects if organisms in the field are able to acclimate to exposures that slowly rise over time. 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 USING WATER OR SEDIMENTS DOWNSTREAM OF MTM-VF

One study (i.e., Merricks et al., 2007) tested media from three streams downstream of MTM-VF in the central Appalachian coalfields. Water and sediment collected from some, but not all, sites downstream of valley fills produced significant toxicity in laboratory organisms.

Water was tested using *C. dubia*. Results were reported as the percent dilution that killed one-half of the test organisms over 48 hours (48-hour LC_{50}). Three streams were tested. The frequency of toxicity was highest in Lavender Fork; undiluted water from three of eight sites sampled 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 *D. 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 treatment 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 mining effluents from Ohio using *C. dubia* and the mayfly *Isonychia bicolor* (Kennedy et al., 2003, 2004, 2005). The effluents 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 *C. dubia* were conducted following EPA protocols and used moderately hard reconstituted water (MHRW)[8] to dilute the effluent. Survival of *C. 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

[8]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).

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 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, *I. 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.

A sodium-dominated effluent from a coal-mine processing impoundment was tested using *C. dubia* and the mayfly *I. bicolor* in 7-day tests using MHRW to dilute the effluent (Echols et al., 2010). Conductivities corresponding to the lowest-observed-effect concentration (LOEC) ranged from 1,508 to 4,101 for *Isonychia* survivorship compared with 2,132−4,250 for *Ceriodaphnia* reproduction. Effects on *Isonychia* reproduction were not studied. 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 might be driving toxicity and how interactions might be producing observed effects. We report on the results of six of these groups of studies. 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-duration toxicity of over 2,900 ion solutions using two crustacean species (i.e., *C. dubia* and *D. magna*) and the fathead minnow (*P. promelas*). *C. dubia* was the most sensitive of the three. The toxicity of ion mixtures varied greatly with composition; total ion concentrations corresponding to acute LC_{50} for *C. dubia* ranged from 390 mg/L to over 5,610 mg/L. For *P. promelas*, LC_{50} 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 from fossil fuel extraction (Tietge et al., 1997). Toxicity of the Ohio coal mine effluent (described above) to *C. 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 *C. dubia* by a factor of about 5 (10% survival predicted vs. 50% survival observed).

5.3.2. van Dam et al. (2010)

In experiments using Australian test organisms and very low calcium water, van Dam et al. (2010) confirmed that Mg^{2+} was a more toxic ion than SO_4^{2-}, and that increasing concentrations of Ca^{2+} reduced Mg^{2+} toxicity. All toxicity tests were conducted using water with much lower Ca^{2+} concentrations than observed downstream of MTM-VF (see Table 2).

5.3.3. Lasier and Hardin (2010)

Lasier and Hardin (2010) developed regression models to predict the toxicity of effluents from anion concentrations and hardness. They tested the toxicity of SO_4^{2-}, Cl^-, and HCO_3^- to *C. dubia* using a three brood reproductive endpoint over 9 days. All of the tests were conducted at lower hardness levels (maximum of 93 mg/L) and higher chloride concentrations (minimum of 85 mg/L) than observed downstream of MTM-VF (see Table 2).

At the highest hardness tested (93 mg/L), LOECs were 1,250 mg SO_4^{2-}/L, 650 mg Cl^-/L, and 450 mg HCO_3^-/L. Concentrations corresponding to a 25% inhibition of reproduction were 1,060 mg SO_4^{2-}/L, 456 mg Cl^- 456/L, and 379 mg HCO_3^-/L. Increasing hardness decreased the toxicity of SO_4^{2-} and Cl^- but had an insignificant effect on the toxicity of HCO_3^-.

5.3.4. 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 LC_{50} value for *C. dubia* was 3,288 mg SO_4^{2-}/L (Soucek and Kennedy, 2005). In all tests, the crustacean *Hyalella azteca* was the most sensitive test organism, followed by *C. dubia*, the bivalve *Sphaerium simile* and the insect *C. tentans*.[9] *H. azteca* was particularly sensitive to SO_4^{2-} at low Cl^- concentrations. At Cl^- concentrations of 1.9 mg/L, *H. azteca* was four times more sensitive to SO_4^{2-} than *C. dubia* (Soucek, 2007b)[10]. 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 as $CaCO_3$.

In three-brood, 7-day tests on *C. dubia*, sublethal effects of SO_4^{2-} occurred at concentrations 2.5 times lower than those that reduced survival (Soucek, 2007a). The LOEC was 899 mg SO_4^{2-}/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 *C. dubia* exposed to 1,000-mg SO_4^{2-}/L.

5.3.5. Meyer et al. (1985)

Meyer et al. (1985) tested four salts using 48-hour tests on *D. magna* and 96-hour tests on *P. promelas*. High hardness dilution water was used (563 mg/L as $CaCO_3$). *D. magna* was more sensitive to all of the salts than *P. promelas*. The relative toxicity of the salts was magnesium sulfate ($MgSO_4$) > NaCl > $NaNO_3$ > Na_2SO_4.[11] The LC_{50} values calculated for $MgSO_4$ were 4,300 mg/L and 7,900 mg/L for *D. magna* and *P. promelas*, respectively. All of these values are well above concentrations reported downstream of MTM-VF (see Table 2).

5.3.6. Skaar et al. (2006)

Skaar and coauthors (2006) conducted acute-duration tests of sodium bicarbonate ($NaHCO_3$) on fathead minnow (*P. promelas*) and pallid sturgeon (*Scaphirhynchus albus*) using test water that simulated conditions in the Tongue and Powder Rivers of Montana. (Limited tests were also conducted using white sucker but are not included in this summary). Hardness values

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[9] *C. tentans* has since been renamed *Chironomus dilutes.*

[10] These SO_4^{2-} results might be unreliable because the synthetic test media contained Cl^- concentrations that, although similar to those observed downstream of MTM-VF, were likely insufficient to maintain healthy *Hyalella* cultures (U.S. EPA 2011).

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

in the tests were not reported. 96-hour LC_{50} values using Powder River water were 840 mg HCO_3^-/L and 1,193 mg HCO_3^-/L for 4-day-old *S. albus* and *P. promelas*, respectively.[12] Earlier life stages appear to be more sensitive: 96-hour exposures to 580 mg HCO_3^-/L resulted in 55% mortality to fathead minnow embryos exposed shortly after fertilization. Longer exposures also appear to increase toxicity. In 30-day tests using fathead minnows, survival declined at concentrations above 290 mg HCO_3^-/L. No decline in growth was noted in surviving fish, however, indicators of kidney damage increased slightly.

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

Evidence from the laboratory toxicity tests suggests that $SO4^{2-}$, HCO^{3-}, Mg^{2+}, and K^+ are the principal contributors to the toxicity of the mixture (see Figure 15). Ca^{2+} in the effluent and receiving water is expected to might reduce the toxicity of the mixture, possibly by mitigating the toxicity of $SO4^{2-}$ and Mg^{2+}. Ionic regulation by organisms depends on the relative proportions of all ions. For this reason laboratory manipulations of one or a few ions at a time are difficult to extrapolate to exposures encountered by organisms in the field. For example, the relatively low concentrations of ions such as Na^+ and Cl^- in effluents downstream of MTM-VF might also be contributing to the overall toxicity of the mixture. Increasing our understanding of the responses of native freshwater organisms to different mixtures of ions and overall ionic strength is a high priority research need (see Section 8).

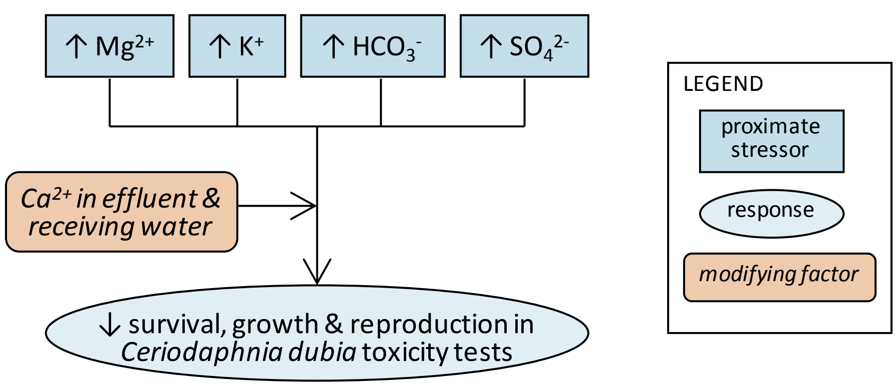

Figure 15. Ions expected to contribute to effects in toxicity tests of water sampled downstream of MTM-VF. See Sections 5.3 and 5.4 for additional details and evidence.

[12] Concentrations reported as $NaHCO_3$ were converted to HCO_3^- concentrations by multiplying them by 0.726.

Applying the Mount et al. (1997) regression models to ion concentrations reported downstream of MTM-VF suggests that the ion mixture at some sites might reach acutely lethal levels to *C. dubia*. The models predict minimal mortality of *C. dubia* (1%) at mean concentrations of each ion summarized in Table 4 (mean specific conductance of 1,023 μS/cm).[13] However, applying the assumption that ion concentrations are strongly correlated, we also calculated predicted toxicity 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 results of the model indicate that SO_4^{2-} concentrations contributed the most to the predicted toxicity, followed by HCO_3^-, Mg^{2+}, and K^+. Cl^- concentrations contributed minimally. The interaction between cations ($Mg^{2+}$and Ca^{2+}) and SO_4^{2-} reduced predicted toxicity substantially. The models predict minimal mortality (1%) for *P. promelas* even at maximum concentrations.

Model predictions of toxicity are generally consistent with the observed *C. 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 summarized in Table 2 (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 conductivity readings (all less than 1,643 μS/cm), only 1 exhibited greater than 50% mortality in the toxicity tests.

Using the anion plus hardness model developed by Lasier and Hardin (2010), *C. dubia* reproduction would be expected to be unaffected even at the highest anion concentrations reported in Table 2. However if, as the authors suggest, hardness does not reduce the toxicity of HCO_3^-, then reproduction would be expected to be 88% of controls at the mean concentration observed and 47% of controls at the maximum concentration of HCO_3^-. The concentrations of HCO_3^- shown in Table 2 reach levels at which effects were observed in chronic tests on the fathead minnow *P. promelas* (Skaar et al., 2006). However, the other ions in the tested mixture were quite different than those reported in Table 2; for example, sodium levels were higher.

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[13]For *C. dubia*, proportion surviving (P) in 48-hour tests was calculated as logit (P) = $\ln[P/(1 - P)] = 8.83 + (-0.0299 \times [K^+]) + (-0.00668 \times [Mg^{2+}]) + (-0.00813 \times [Cl^-]) + (-0.00439 \times [SO_4^{2-}]) + (-0.00775 \times [HCO_3^-]) + (-0.446 \times 2) + (0.00870 \times 2 \times [K^+]) + (0.00248 \times 2 \times [Cl^-]) + (0.00140 \times 2 \times [SO_4^{2-}])$ (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 *P.promelas,* proportion surviving (P) in 96-hour tests was calculated as logit (P) = $\ln[P/(1 - P)] = 4.70 + (-0.00987 \times [K^+]) + (-0.00327 \times [Mg^{2+}]) + (-0.00120 \times [Cl^-]) + (-0.000750 \times [SO_4^{2-}]) + (-0.00443 \times [HCO_3^-])$ (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.

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 *I. bicolor* and the amphipod *H. 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 younger organisms might be more sensitive. The concentrations associated with effects from the sodium-dominated mine effluent tested using the mayfly *Isoynchia* were lowest in tests using the smallest and presumably youngest organisms (Echols et al., 2010). In tests with bicarbonate, 7-day-old *H. azteca* were two times more sensitive than 14-day old organisms (Lasier et al., 1997). In studies of metal salts (Cu, Cd, and Zn), in experimental streams, (Kiffney and Clements, 1996) toxicity increased as organism size decreased. Just-fertilized embryos of *P. promelas* were about 1.5 times more sensitive to sodium bicarbonate than 4-day old larvae (Skaar et al., 2006).

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 (see Figure 16). 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-duration 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.

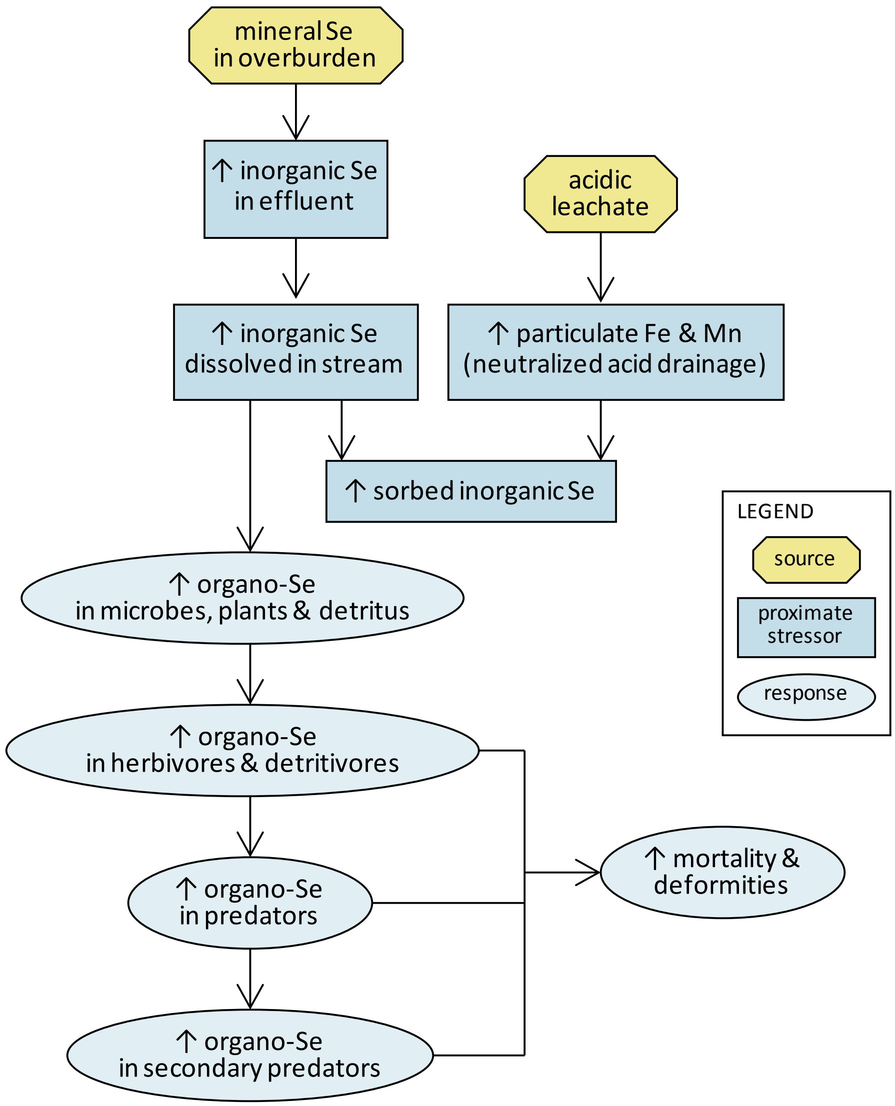

Figure 16. Selenium transformation, transfer and effects expected in aquatic ecosystems downstream of MTM-VF. See Section 5.5.1 for additional details and evidence.

5.5.1.1. *Selenium Dynamics in Aquatic Ecosystems*

The complex dynamics of Se have been recently summarized by Chapman et al. (2010), Luoma and Presser (2009), and Presser and Luoma (2010). Selenium, leached from coal and organic overburden, enters streams in valley fill effluents (see Section 4.3.1). Dissolved oxy anions of selenate (Se^{+4}) and selenite (Se^{+6}) are taken up by microbes, algae, and plants and converted to organic forms. In the streams like those below MTM-VF, the primary community that can perform this conversion is the periphyton growing on rocks and woody debris, and the conversion rates are relatively low. However, uptake, conversion, and retention of Se are more efficient in lentic systems such as the reservoirs that occur downstream in some watersheds. Alternatively, in streams with iron oxyhydroxides or manganese dioxide due to neutralization of acidic leachates, significant sorption of Se to those amorphous minerals might occur. Herbivores and detritivores (largely aquatic macroinvertebrates) accumulate Se by grazing and collecting organic particles. These primary consumers are in turn consumed by predators including fish, amphibians, and birds. Secondary predators, such as largemouth bass or herons, can further accumulate Se, but such large predators occur primarily in larger water bodies. In sum, Se can bioaccumulate and even biomagnify where retention is high and food chains are long. In the region of concern, these conditions could occur in reservoirs and potentially in riparian wetlands. Se bioaccumulation is expected to be lower in the streams below MTM-VF operations. However, these conclusions are based on general knowledge of Se dynamics and not on specific studies in the region of concern.

5.5.1.2. *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), which are similar to the concentrations observed downstream of MTM-VF. 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 oligochaete 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.

5.5.1.3. *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 waters had concentrations >5 mg/kg, putting them at "moderate hazard" for toxic effects based on the scale developed by Lemly (1993).

5.5.1.4. *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 Louisiana waterthrushes occur more commonly in the area of concern and forage on aquatic invertebrates, so they 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) could be at risk from Se-contaminated fish (see Section 5.5.1.3). The 10^{th} 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

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 *C. dubia* in hard water (hardness = 184 mg/L) yielded a mean 48-hour LC_{50} of 15.2 mg Mn/L for *C. dubia* and a 96-hour LC_{50} value for *H. azteca* of 13.7 mg Mn/L (Lasier et al., 2000). In 7-day tests, *C. dubia* reproduction (number of young per female) was inhibited 50% at mean concentrations of 11.5 mg Mn/L. In 62-day life-cycle tests using brown trout, concentrations associated with a 25% inhibition in survival or growth were 5.59 mg/L and 8.68 mg/L at hardness levels of 150 and 450 mg/L, respectively (Stubblefield et al., 1997).

In a study of biochemical effects, concentrations of chemicals involved with cellular redox regulation were reduced at concentrations lower than reported by Pond et al. (2008):

glutathione levels were reduced in caddisflies (*Hydropsyche betteni*) exposed to 0.05 mg Mn/L, and cysteine levels were reduced in mayflies (*Maccaffertium modestum*) exposed to 0.10 mg Mn/L (Dittman and Buchwalter, 2010). However, no overt toxic effects were reported in this study, or in companion bioaccumulation tests that exposed a wide variety of Appalachian stream insects to concentrations up to 0.40 mg Mn/L (Dittman and Buchwalter, 2010).

Maximum concentrations of total and dissolved Fe of 0.65 mg/L and 0.28 mg/L, respectively, have been observed downstream of MTM-VF (see Table 2). Iron toxicity decreases as pH increases (see the recent review by Phippin et al. [2008]). The pH of water downstream of MTM-VF ranged from 6.3−8.9 (see Table 2). Observed iron concentrations are similar to the 96-hour tolerance limit concentration of 0.32 mg $FeSO_4$/L at pH 7.5 reported for *Ephemerella* sp. survival in a study conducted prior to standardized toxicity test protocols (Warnick and Bell, 1969). Other organisms tested in that study were less sensitive. LC_{50} values were 16.0 mg Fe/L for both the stonefly *Acroneuria lycorias* and the caddisfly *Hydropsyche betteni* in 9- and 7-day tests, respectively. No differences in survival, feeding, or escape activity were observed in experiments exposing field-collected larval mayflies (*Leptophlebia marginata*) to up to 50 mg Fe/L at pH 7 for about 30 days (Gerhardt, 1992). The effect concentration for 50% of the tested organisms (EC_{50}) for reduced escape activity was calculated as 70 mg Fe/L at circumneutral pH (between 5.95 and 6.74) (Gerhardt, 1994). In studies using *D. magna* at pH of 7.5, reproduction declined 16% at 4.38 mg Fe/L (Biesinger and Christensen, 1972).

Observed iron concentrations reach levels that exceed 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, FEC_{20}s) corresponded to a 20% decline in the organism numbers compared with reference sites and were estimated from the 90^{th} 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 might 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 below MTM-VF. 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

Table 11. Field-based 20% effect concentrations (FEC_{20}) for iron (Linton et al., 2007)

Macroinvertebrate family	FEC_{20} (mg total iron/L)
Leptophlebiidae	0.21
Emphemerellidae	0.43
Philopotamidae	0.44
Psephenidae	0.48
Heptageniidae	0.66
Elmidae	1.13
Baetidae	1.48
Tipulidae	7.05

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 probable effects levels (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.

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.

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.They are directly relevant to both the exposures and biota of interest. Macroinvertebrate and fish assessments consistently indicate degraded biological conditions downstream of MTM-VF.

6.1. EFFECTS ON BIOLOGICAL COMPOSITION

Mountaintop mining and associated valley fills in a watershed are associated with degraded community compositions 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; Armstead et al., 2004; Kirk and Maggard, 2004; Hartman et al., 2005; Merricks et al., 2007; Pond et al., 2008). Although a number of different biological responses have been associated with the effects of MTM-VF (see Figure 17), all the relevant studies reviewed found that mayfly (i.e., the 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 (WVSCI) 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) found that benthic macroinvertebrate indices were lower in streams with upstream valley fills. With the exception of the fall of 2000,

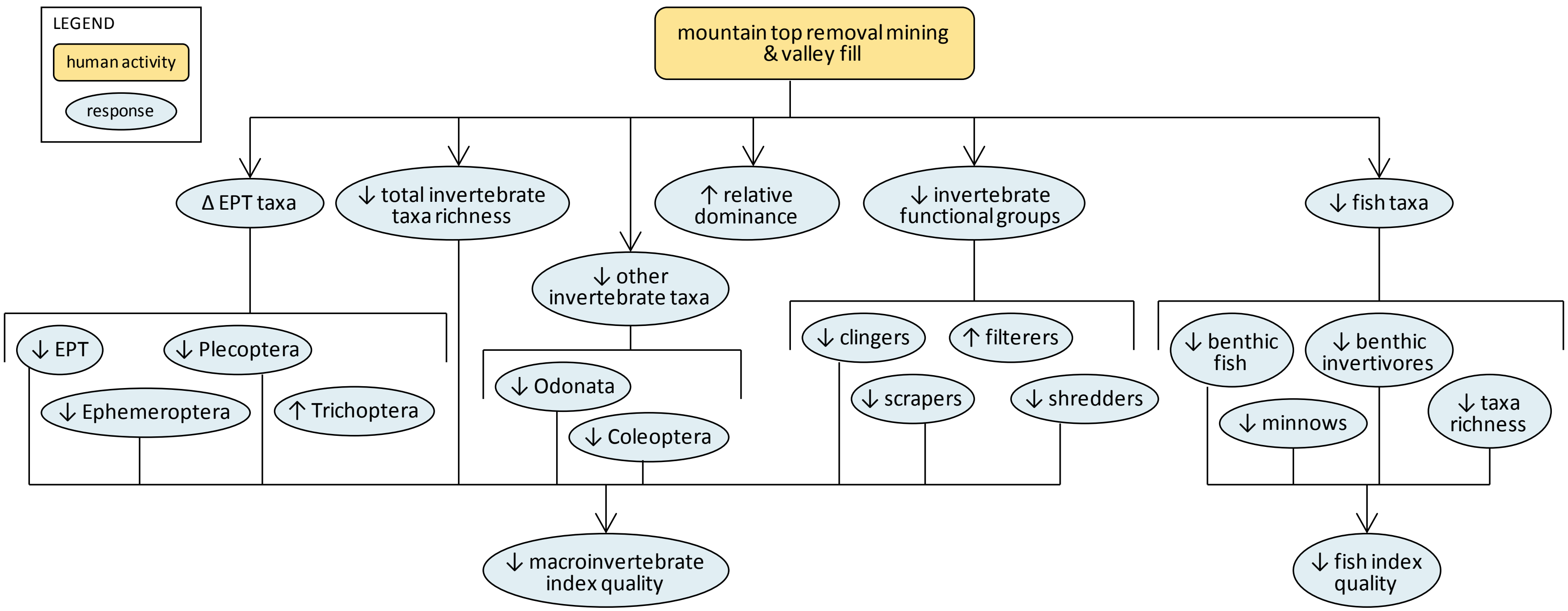

Figure 17. Macroinvertebrate and fish responses associated with MTM-VF. Responses were significantly different from reference sites in at least one of the reviewed studies. Both negative and positive responses of non-insects and midges were observed and are not shown. See Section 6.1 and Table 13 for additional details and evidence.

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 survey comparing MTM-VF streams (n =17) to unmined reference streams (n = 14)	Fish IBI	Lower
		Benthic invertivores	Lower
		Native Cyprinidae (minnows) richness	Lower
		% gravel spawners	No difference
		% predators	No difference
		Intolerant species richness	No difference
		% nonnative fish	No difference
		% macro-omnivores	No difference
		% tolerant species	No difference
		% Cottidae (sculpins)	No difference
	Benthic macroinvertebrate surveys (spring) comparing MTM-VF streams (n = 9 in 1999, n = 10 in 2000) to unmined reference streams (n = 15)	Invertebrate IBI	Lower
		Total taxa richness	Lower
		EPT taxa richness	Lower
		Hilsenhoff Biotic Index	No difference
		% 2 dominant taxa	No difference
		% EPT taxa	No difference
		% Chironomidae	No difference
Hartman et al., 2005	Benthic macroinvertebrate survey comparing streams with valley fills (n = 4) to reference streams without valley fills (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

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]
Armstead et al., 2004	Benthic macroinvertebrate survey comparing MTM-VF streams (n = 14, winter; n = 15, spring) to streams in watersheds without mining activity (n = 9, winter; n = 10, spring)	Total density	No difference
		Taxa richness	No difference
		Hilsenhoff's Biotic Index	Higher
		EPT density	No difference
		EPT richness	Lower
		% EPT	Lower
		% 2 dominant taxa	Increase
		% Chironomidae	No difference
		% Ephemeroptera	Decrease
		% Plecoptera	Decrease
		% Trichoptera	No difference
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 streams with valley fills (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
Pond, 2010	Benthic macroinvertebrate survey comparing MTM-VF streams (20) to unmined reference streams (44)	Ephemeroptera richness	Lower
		% Ephemeroptera	Lower
Stauffer and Ferreri, 2002	Fish communities were compared in streams with valley fills (n = 9) to streams 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 value 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.

IBI = Index of Biotic Integrity; MBI = Macroinvertebrate Bioassessment Index.

the macroinvertebrate index was significantly different among stream classes, and the 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 WVSCI scores ranging 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) used the Kentucky Macroinvertebrate Bioassessment Index (MBI) (Pond and MacMurray, 2002), which includes four components of macroinvertebrate community condition. 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"). In a time-series study (i.e., April and

October samples from 1996 to 2003) of a third-order stream affected by two smaller tributaries with MTM-VF, Kirk and Maggard (2004) generally observed WVSCI scores in the “fair” to “good” range at the downstream site compared to scores in the “good” to “very good” range at an upstream reference site, after a period of drought from fall 1998 through spring 1999. During the early part of the study (i.e., 1996 to 1998), the WVSCI scores were similar between the sites in the “good” to “very good” range.

Merricks et al. (2007) found a HBI score of 1.91 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. A similar pattern of increased HBI was observed during sampling in two seasons by Armstead et al. (2004).

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) and a family-level biotic index (WVSCI, 3.4 vs. 4.3). Using the genus-level index, 93% (25 of 27) of sites downstream of mining activity exhibited scores indicative of biological impairment, compared with none (0 of 10) of the sites that were in reference (unmined) watersheds. Using the family-level index, 63% (17 of 27) of downstream of mining activity exhibited scores indicative of biological impairment, compared with none (0 of 10) of sites that were in reference (unmined) watersheds (Pond et al. 2008).

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), in West Virginia (mean generic richness of 21.7 at mined sites and 31.9 at unmined sites, Pond et al., 2008), and in a combination of sites from both states (median family richness of 12−13 at sites with fills and 18−21 at unmined sites in spring, Fulk et al., 2003). In a time-series study (i.e., April and October samples from 1996 to 2003) of a third-order stream affected by two smaller tributaries with MTM-VF, Kirk and Maggard (2004) generally observed lower macroinvertebrate richness at the downstream site compared to an upstream reference site, particularly since 1999. In contrast, Armstead et al. (2004) and Merricks et al. (2007) found no significant difference in taxonomic richness between streams with valley fills and streams without valley fills in the watershed.

6.1.1.3. *Benthic Macroinvertebrate Density*

No difference was found in the total density of macroinvertebrates between streams with valley fills and reference streams (Armstead et al., 2004; Hartman et al., 2005). In a time-series study (i.e., April and October samples from 1996 to 2003) of a third-order stream affected by two smaller tributaries with MTM-VF, Kirk and Maggard (2004) generally observed greater macroinvertebrate densities at the downstream site compared to an upstream reference site early in the study (i.e., 1996−2000) but lower densities in the later part of the study (2000−2003).

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; 11% in streams with valley fills and settling ponds; 22% in unmined streams, Armstead et al., 2004) 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 streams with valley fills in the watershed than it was in streams without valley fills (Hartman et al., 2005). The percentage of the community as collector-filtering macroinvertebrates (i.e., organisms that feed on suspended particulate organic matter, including algae) was greater in streams downstream from both the fills and settling ponds (Merricks et al., 2007; Armstead et al., 2004), 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

The only study that included Coleoptera 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 effects of MTM-VF on Diptera population characteristics were 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). Armstead et al. (2004) and 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. In another study, the percent Chironomidae was greater 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), but in a time-series study (i.e., April and October samples from 1996 to 2003) of a third-order stream affected by two smaller tributaries with MTM-VFs, percent Chironomidae did not consistently increase or decrease at the downstream site compared to the upstream reference site. 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 might account for the equivocal results.

6.1.1.5.3. Ephemeroptera

Ephemeroptera population characteristics showed the most definitive changes associated with mining activities, being 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). Pond (2010) found decreases in the abundances of individual mayfly genera, such as *Ameletus*, *Drunella*, *Ephemerella*, *Cinygmula*, *Epeorus*, and *Mccaffertinum* in mine-impacted streams. 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. Four additional studies report similar observations of proportion. Merricks et al. (2007) found 3% Ephemeroptera in streams with mountaintop mining in the watershed and 17% in reference streams in West Virginia. Pond et al. (2008) found 7% Ephemeroptera in streams with mountaintop mining in the watershed and 45% in streams with no mining in West Virginia. Pond (2010) found 2% Ephemeroptera in streams with mountaintop mining in the watershed and 45% in streams reference streams in Kentucky. Armstead et al. (2004) found 4% (winter) or 16% (spring) Ephemeroptera in streams with valley fills and 30% (winter) or 42% (spring) Ephemeroptera in reference streams in West Virginia. In a time-series study (April and October samples from 1996 to 2003) of a third-order stream affected by two smaller tributaries with MTM-VF, Kirk and Maggard (2004) found reduced % Ephemeroptera at the downstream site compared to an upstream reference site, particularly in the years after mining ended. Likewise, Ephemeroptera richness was significantly lower in mine-impacted streams (Merricks et al., 2007; Pond et al., 2008; Pond, 2010). Also, using nonmetric scaling ordination and a nonparametric multiresponse permutation procedure, Pond (2010) found that Ephemeroptera assemblages in reference streams were significantly dissimilar (i.e., the assemblages were not the same) from those in mine-impacted streams.

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

The evidence for MTM-VF Plecoptera populations is weaker. In studies of taxa richness, Plecoptera richness was lower in streams with mining in the watershed (2.7 genera) than in streams without mining (6 genera) (Pond et al., 2008). Another study found 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 in studies of relative abundance (i.e., percent Plecoptera). Merricks et al. (2007) found lower Plecoptera percentages in sites downstream of valley fills (4.5% in mined streams and 52% at a reference site). Armstead et al. (2004) found decreased percentages in streams with valley fills sites in spring (11% in valley fills streams and 21% in reference streams) but found no difference in winter. Pond et al. (2008) did not detect a difference in percent Plecoptera 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 stream reach just downstream of the settling pond was sampled, the proportion of the macroinvertebrate assemblage that was in the order Trichoptera was greater than the stream reaches upstream of the pond and downstream of the valley fill or streams without mining (20% in mined streams just downstream of the settling pond, 4.1% in mined streams upstream of the settling pond, and 3.7% in reference watersheds, Merricks et al., 2007). As the distance downstream of the settling pond increased, the proportion of the macroinvertebrate assemblages that was Trichoptera decreased.

Two other studies found no difference among streams downstream of fills and those without fills (Armstead et al., 2004; Hartman et al., 2005).

6.1.1.5.7. Ephemeroptera, Plecoptera, and Trichoptera (EPT) indices

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 two studies (Pond et al., 2008: EPT generic richness of 17.9 at unmined sites and 8.9 at filled sites; Armstead et al., 2004: EPT taxa richness of 9 at unmined sites and 6 at filled sites; Fulk et al., 2003: EPT family richness of 12−13 at unmined sites and 9 at filled sites in spring) and mixed results (decreased EPT richness in two valley fill streams and no differences in two other valley fill streams) in another (Merricks et al., 2007). Hartman et al. (2005) observed no differences 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 (Armstead et al., 2004; Merricks et al. 2007; Pond et al. 2008). In a time-series study (i.e., April and October samples from 1996 to 2003) of a third-order stream affected by two smaller tributaries with MTM-VF, percent EPT was generally lower at the downstream site compared to an upstream reference site, particularly in the period since 1999 (Kirk and Maggard, 2004).

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 and among insect orders. Particularly important in these effects are taxonomic shifts because of differing sensitivity among the three orders: Ephemeroptera, Plecoptera, and Trichoptera. As described previously Plecoptera and Trichoptera, in general, do not appear to be as sensitive to the effects of MTM-VF as Ephemeroptera. These differences in sensitivity have been observed for other stressors, such as metals and low pH (Griffith et al., 1995; Luoma et al., 2010).

6.1.1.5.8. 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). 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). However, bioassessment surveys, such as Hartman et al. (2005), Howard et al. (2001), and the other studies discussed previously, do not generally sample one group of noninsect benthic macroinvertebrates, river mussels of the family Unionidae, because of their highly clumped distribution in stream systems (Neves and Widlak, 1987). Many river mussels are already threatened or endangered, because of sedimentation, construction of dams, and other alterations of rivers of this region and elsewhere (Layzer et al., 1993; FWS, 2004).

6.1.1.5.9. Macroinvertebrate taxa dominance

Dominance metrics generally measure shifts in relative abundance from more sensitive species to more tolerant species. Dominance of a community by a few organisms is expected to increase with stress (e.g., Yuan and Norton, 2003). Armstead et al. (2004) observed that the percentage of the macroinvertebrate assemblage that was the two most numerically dominant taxa increased in streams with valley fill (i.e., 65 to 67%) as compared with reference streams (i.e., 50 to 54%).

6.1.2. Fish

Fish community attributes, such as 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) indicates 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). Ferreri et al. (2004) conducted another study that compared stream reaches downstream of valley fills in the Mud River basin (i.e., 8 sites) with stream reaches without mining in the Big Ugly Creek basin (i.e., 5 sites), both tributaries to the Guyendotte River of southwestern West Virginia and found the same pattern of fewer total fish species (i.e., median = 17 in unmined reaches versus median = 8 in mined reaches) and fewer benthic fish species (i.e., median = 6 in unmined reaches versus median = 1.5 in mined reaches).

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 (i.e., 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) found the reduced index score was caused by fewer minnow species (median = 5.0 in unmined versus median = 4.3 in streams with valley fills) and benthic insectivores (median = 6.0 in unmined versus median = 4.9 in streams with valley fills) 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).[14]

6.1.3. Amphibians, Particularly Salamanders

It is well-known that salamanders are an important part of the aquatic vertebrate assemblage in the central Appalachians (Davic and Welsh, 2004), particularly in the small intermittent and permanent streams impacted by MTM-VF. Despite this and the suggestion that salamanders be used for biomonitoring elsewhere (Welsh and Ollivier, 1998; Welsh and Droege, 2001; Ohio EPA, 2002), only one field study has been conducted to study the effects on salamanders, and most field studies have concentrated on the more commonly used fish and macroinvertebrates (see above). Williams and Wood (2004) quantified salamander diversity and abundance in four second- and third-order reaches downstream from valley fills and in four intermittent, first- and second-order reference reaches that were unimpacted by MTM-VF.

[14]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.

While overall species richness was similar in the streams (i.e., 7 species in valley fill streams versus 8 species in reference streams), mean salamander abundance per 35-m stream reach was greater in the reference streams (i.e., 25.7 ± 14.4 [mean ± SE], 8−66 [range]) than in the valley fill streams (15.9 ± 9.5, 0–76). Salamander abundance was particularly correlated with the number of rocks in the stream reach ($r = 0.63$, $p < 0.001$). Using leaf litter bags, 20 salamander larvae were captured in the reference streams versus only 3 salamander larvae in the valley fill streams.

6.2. EFFECTS ON ECOLOGICAL FUNCTION

To date, few studies have assessed the impact of MTM-VF on ecological function (e.g., biogeochemical cycling) in downstream habitats. One study, Fritz et al. (2010), compared in-channel, standing crop biomass of coarse benthic organic matter (CBOM) and breakdown rates and invertebrate colonization of oak leaf litter in channels associated with valley fills with that in natural, forested streams unaffected by MTM-VFs. Three classes of channels or streams in terms of flow duration were identified: ephemeral or intermittent channels constructed on valley fills or permanent channels downstream of valley fills. They found that standing crop biomass of CBOM was similar between channels associated with valley fills and natural streams in the permanent and intermittent reaches, but CBOM was greatest in the natural ephemeral streams and least in the constructed ephemeral channels on the valley fills. Leaf litter breakdown rates, whether per day or per degree day, were similar in both types of ephemeral channels (i.e., on valley fills versus natural). In intermittent or perennial channels, leaf litter breakdown rates were greater in natural, forested stream than in channels on or downstream from valley fills. The densities of all invertebrates or just invertebrate shredders on the leaf packs were also greater in the intermittent or perennial natural forested streams than in the intermittent channels on the valley fills or the permanent channel downstream from the valley fill. Because differences in water temperature among the sites are removed by the use of cumulative degree-days and the differences in flow duration are partitioned among the three classes of channels, the differences in leaf breakdown rates appear to be most related to conductivity. Beyond this study, 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 18). 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 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.

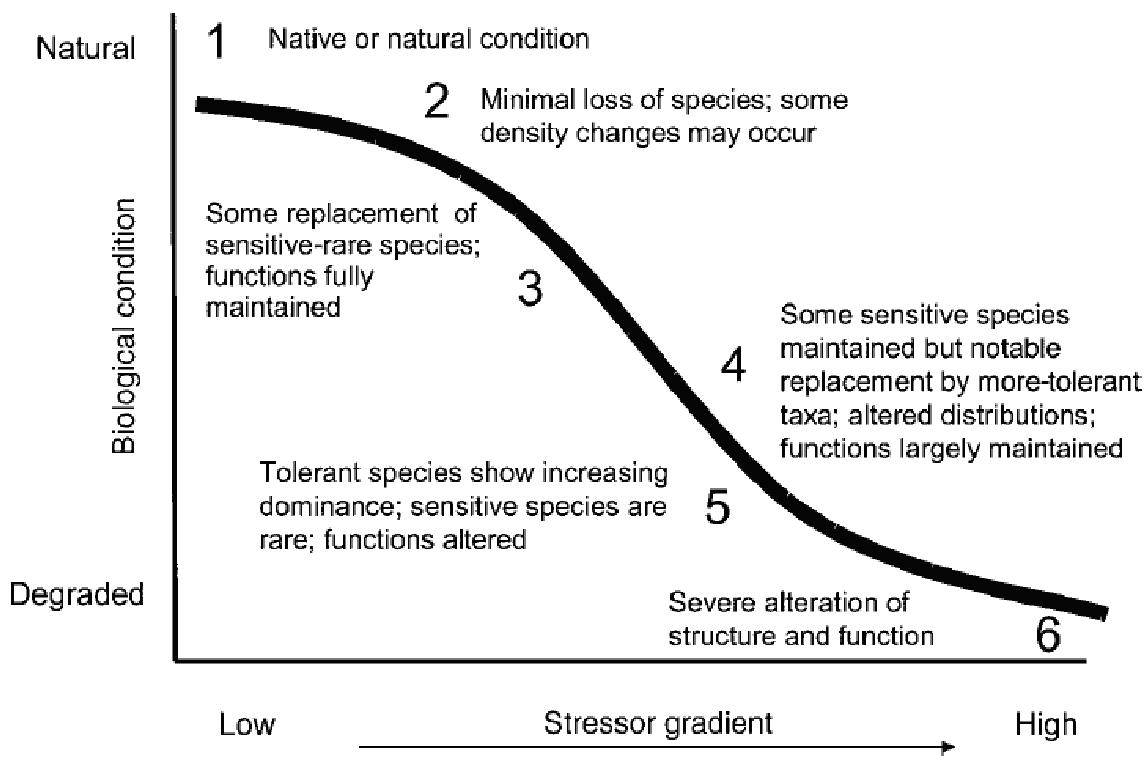

Figure 18. Conceptual model depicting stages of change in biological conditions in response to an increasing stressor gradient.

Source: Davies and Jackson (2006). Used with permission from the publisher.

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 (see Figure 19).

6.4.1. Ion Concentration

All studies report elevated ion concentration in MTM-VF (see Table 14), and most show strong negative 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; Pond, 2010; Timpano et al., 2010).

Several studies of other types of mining discharges have reported associations with conductivity. Conductivities ranging from 500–8,000 µS/cm had a significant negative correlation with the number of pollution sensitive taxa in benthic macroinvertebrate assemblages (Soucek et al., 2000; Kennedy et al., 2003). In a study that followed effects downstream of an alkaline mine discharge in a tributary of the Monongahela River in southwestern Pennsylvania (Kimmel and Argent, 2010), fish species richness and total abundance declined with increases in conductivity, although effects based on comparing species dissimilarity of the assemblage to a reference site (*sensu* Courtemanch and Davies, 1987) were most apparent only when conductivity levels exceeded 2,300 µS/cm (Kimmel and Argent, 2010). However, the authors also state that tributaries of the Monongahela River, in general, support a relatively pollution-tolerant fish assemblage (i.e., only 2% classified as intolerant) because of the historical impacts of coal mining, sewage discharges, agriculture, and urbanization (Kimmel and Argent, 2006). 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 noted that 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 WVSCI ($r = -0.80$), total generic richness ($r = -0.74$), EPT generic richness ($r = -0.88$), number of Ephemeroptera genera ($r = -0.90$), the number of Plecoptera genera ($r = -0.75$), and percent Ephemeroptera ($r = -0.88$). Of the sites having greater than 500 µS/cm specific conductance, 100% (20 of 20) had genus-level macroinvertebrate index scores indicative of

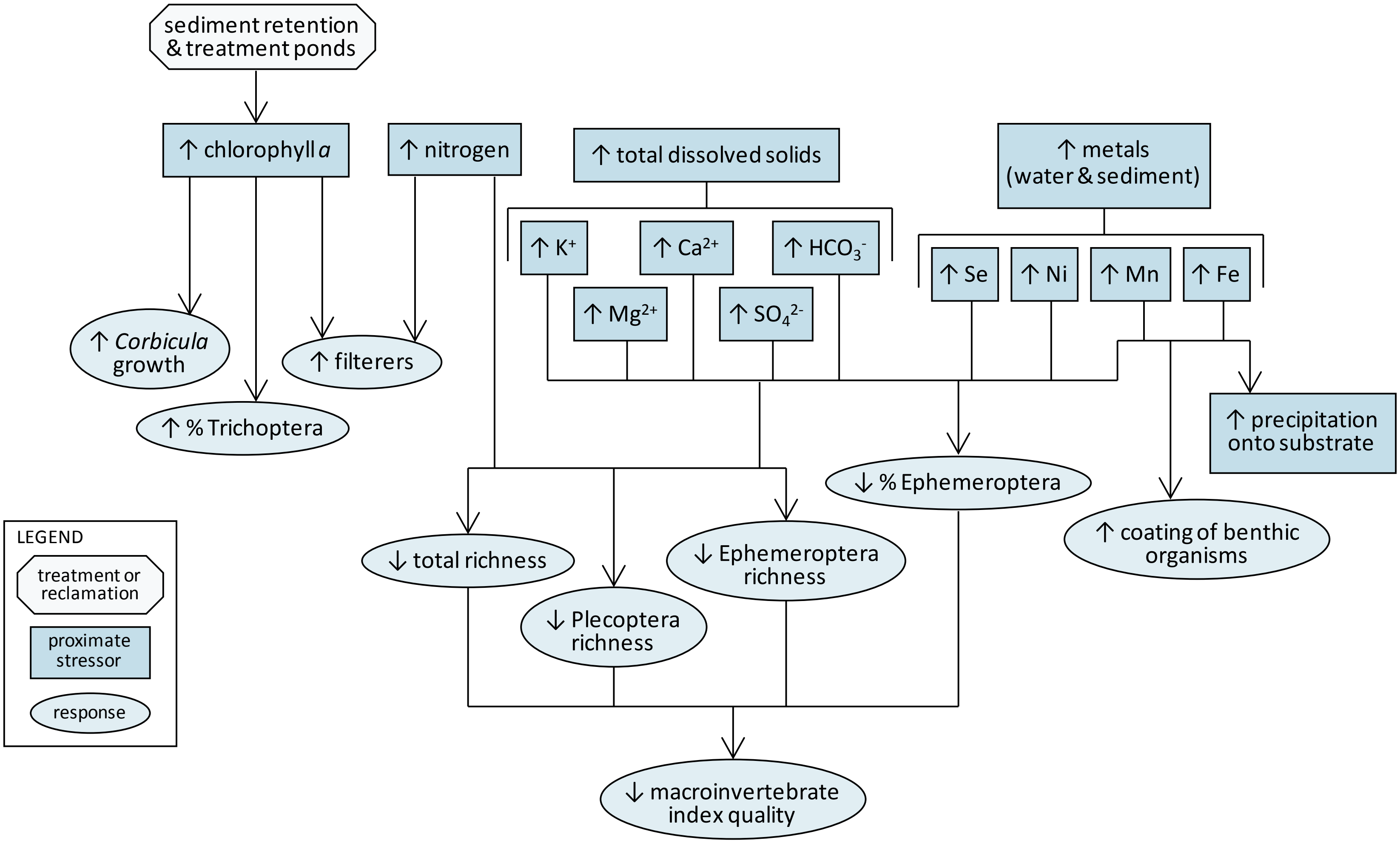

Figure 19. Macroinvertebrate and fish responses associated with stressors and treatment ponds in studies of MTM-VF. Responses shown were observed in at least one of the reviewed studies. Manganese and iron precipitates were directly observed. Other responses were either significantly different from sites upstream of ponds or were strongly correlated with the stressor (i.e., absolute correlation coefficient >0.7). See Section 6.4 for additional details and evidence.

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	1,051	4	150
Howard et al., 2001	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)
Pond, 2010	µS/cm	20	940 (161–2,390)	44	51 (16–159)
Stauffer and Ferreri, 2002[c]	µmhos/cm	8	1,716 (513–2,330)	9	164 (125–210)
Timpano et al., 2010	µS/cm	17	13[d] (25–970)	3	NR[d]

[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.
[d]The median value and not the mean was reported for the mined sites. No values were reported for the reference sites.

biological impairment; 85% (17 of 20) had family-level macroinvertebrate index scores indicative of biological impairment (Pond et al. 2008).

In an analysis of streams downstream from mining and valley fills in Virginia, Timpano et al. (2010) observed strong negative relationships between specific conductance and the biotic metrics: EPT taxa richness ($r = -0.76$), Ephemeroptera taxa richness ($r = -0.71$), Plecoptera taxa richness ($r = -0.72$), total taxa richness ($r = -0.50$), and collector taxa richness (−0.58). Also, a metric that increases with impairment, percent abundance of five most dominant taxa, exhibited a positive relationship with specific conductance ($r = 0.64$, Timpano et al., 2010). In an analysis of Kentucky streams, Pond (2010) found specific conductance to be negatively correlated with percent abundance of Ephemeroptera ($r = -0.72$). Using a time-series data set (i.e., April and October samples from 1996 to 2003) from just two sites in a third-order stream upstream and downstream of its confluences with two smaller tributaries with MTM-VFs, Kirk and Maggard (2004) found a weaker negative correlation with specific conductivity and WVSCI scores ($r = -0.34$) but a strong negative correlation with percent abundance of Ephemeroptera ($r = -0.71$).

Pond et al. (2008) further demonstrated a decline in number of Ephemeroptera taxa and their proportion of the assemblage when conductivity levels exceeded approximately 500 μS/cm. Using a nonparametric changepoint analysis using the deviance reduction method, Pond (2010) found a threshold-type response for percent abundance of mayflies centered on a specific conductance of 175 μS/cm (confidences limits: 124−336 μS/cm). TDS or specific conductance also had strong negative correlations with biological metrics (Stauffer and Ferreri, 2002; Pond et al., 2008; Timpano et al., 2010). Although the strength of correlations of richness metrics and macroinvertebrate indices with individual ions varied somewhat, strong negative correlations (i.e., absolute value of $r > 0.7$) were found in at least one of these studies with HCO_3^-, Ca^{2+}, SO_4^{2-}, Mg^{2+}, and K^+. 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.

Additional insights on possible mechanisms can be found in the physiological literature on osmoregulation (e.g., Bradley, 2009). Although earlier literature emphasized regulation of osmotic pressure and cell volume as mechanisms by which salts affect freshwater organisms (e.g., Kapoor, 1979; Pierce 1982), regulation of internal ionic concentrations has been emphasized more recently. Freshwater vertebrates, including fish and amphibians, as well as invertebrates, such as aquatic insects, crayfish, and unionid mollusks, are invariably hyper-regulators (Kirschner, 1970; Dietz and Branton, 1975; Dietz, 1979; Goss et al., 1992; Harvey, 1992; Henry and Wheatly, 1992; Cooper, 1994; Wheatly and Gannon, 1995; Ehrenfeld and Klein, 1997; Perry, 1997; Kirschner, 2004). These animals maintain internal concentrations of ions, such as Na^+, K^+, Cl^-, Mg^{2+}, and SO_4^{2-}, that are greater than the concentrations of these ions in unimpaired freshwaters. These ions are moved into the organism against concentration gradients using several mechanisms. In particular, H^+ (or combined with ammonia to form NH_4^+) and HCO_3^- are by-products of respiration and are exchanged for Na^+ and Cl^-, respectively, to move these ions into the organism (Evans, 1975; Dietz, 1979; Grosell et al., 2002; Kirschner, 2004). In experiments with goldfish (*Carassius auratus*), the addition of HCO_3^- to the external medium inhibited the uptake of Cl^- (Maetz and Garcia Romeu, 1964). High concentrations of HCO_3^- downstream of valley fills might similarly be inhibiting uptake of Cl^- and export of HCO_3^-. In addition to reducing internal Cl^- concentrations, the excess internal HCO_3^- might also alter the acid-base balance within the organisms (Goss et al., 1992; Henry and Wheatly, 1992).

6.4.2. Specific Metals and Selenium

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 correlations (*r* ranged from −0.70 to −0.98) between macroinvertebrate metrics and metals. For example, Ephemeroptera richness was negatively correlated with Fe, Mn, and Ni; EPT taxa richness was negatively correlated with Mn and Ni. 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. Ephemeroptera and Plecoptera richness were both strongly and negatively correlated with Se concentrations (Pond et al., 2008). These results suggest that elevated metal concentrations associated with mine-impacted streams might 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 (see Figure 19). Pond et al. (2008) found strong correlations (i.e., absolute Spearman correlation coefficients greater than 0.7) between $NO_2 + NO_3$ and the number of Ephemeroptera taxa; Plecoptera taxa; the sum of Ephemeroptera, Plecoptera, and Trichoptera taxa; and the sum of all taxa. In addition, the relative abundance of Ephemeroptera was strongly and negatively correlated with $NO_2 + NO_3$. However, total phosphorus levels were below detection limits at all sites. Merricks et al. (2007) evaluated changes in the composition of the structural and functional composition of the macroinvertebrate assemblages downstream of the settling ponds in mined streams along with growth of the filter-feeder, *Corbicula* (Asiatic clam) to assess potential organic enrichment. Stations just downstream of the ponds had significantly greater proportions of collector-filterers compared to stations upstream of the ponds, and these proportions decreased further downstream. Similar patterns in the proportion of the macroinvertebrate assemblage that were Trichoptera and in the growth of *Corbicula* were observed. Moreover, water column chlorophyll *a* concentrations were generally high in the settling ponds, indicating the presence of algae in the pond that is food for filter-feeders like *Corbicula* and many Trichoptera species (e.g., the Hydropsychidae). 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).

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. As discussed in Section 6.1.1.4, decreases in macroinvertebrates that cling to rocks (clingers) were observed in one study (Howard et al., 2001). Similar decreases have been associated with increases in fine sediments in a regional study, suggesting that the observed declines might be associated with changes in sediments. (Pollard and Yuan, 2009). However, the field studies we reviewed found no systematic relationship between macroinvertebrate metrics and habitat assessment measures including measures of fine sediments and turbidity (Howard et al., 2001; Hartman et al., 2005; Merricks et al., 2007), which might suggest that habitat characteristics were not all that different between reference and mined stream sites. Total rapid bioassessment procedure habitat scores were correlated with macroinvertebrate indices in the study by Pond et al. (2008). However, individual subscores show only weaker correlations, and the investigators did not observe excessive sedimentation in sites downstream of valley fills.

Iron can precipitate out of the water onto organisms and substrates, clogging gills and degrading habitat by cementing together sediment particles and promoting the growth of Fe-depositing bacteria (Vouri, 1995). Fritz et al. (2010) reported the presence of $FeSO_4$ precipitates and Fe-depositing bacteria at all study sites downstream of MTM-VF; neither was observed at reference sites. Pond (2004) noted the presence of both Mn and Fe precipitates on organisms downstream of MTM-VF (see Figure 20). We did not find any studies documenting the types or prevalence of precipitates downstream of MTM-VF or any studies that distinguished the effects of these precipitates from water quality impacts.

Figure 20. Mn (black) and Fe (orange) deposits on a caddisfly collected downstream of a mountaintop mine and valley fill.

Source: Pond (2004).

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 could influence macroinvertebrate composition in downstream habitats. Headwater streams are critical to downstream ecological condition and their alteration, as in mountaintop mining and valley fill activities, could impact the integrity and the sustainability of downstream habitats. As reviewed in Section 3.4, 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 can 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).

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 could 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.

7. RECLAMATION, MITIGATION, AND RECOVERY

In the following section, we address reclamation and mitigation efforts following MTM-VF (see Figure 21). In particular, we examine (1) the dominant post-SMCRA reclamation practice of seeding with grasses; (2) the Forestry Reclamation Approach; (3) the creation of channels on valley fills; (4) the use of erosion control structures and creation of wetlands; and finally (5) enhancing stream structural heterogeneity and riparian areas below the valley fill. We discuss these practices as they relate to streamflow, water quality, and aquatic communities. This discussion is limited to on-site reclamation and mitigation techniques. For a discussion of off-site mitigation efforts, see Chapter III and Appendix D of the PEIS (U.S. EPA, 2003, 2005).

7.1. RECLAMATION OF MTM-VF SITES

7.1.1. Overview

Prior to SMCRA (30 U.S.C. § 1231, *et seq*.) passed in 1977, mining practices left large areas of unstable land and eroding hillslopes that impaired streams and created human health risks from mudslides and pollution. That history, plus 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 some of the productivity and ecological function of native forests. These techniques, termed the Forestry Reclamation Approach (FRA), are based in part on research and extension efforts of the Appalachian Regional Reforestation Initiative, a coalition of groups formed to promote reforestation of Eastern coal mine sites (http://arri.osmre.gov) (Skousen et al., 2009).

Under SMCRA, reclamation is not considered complete until water quality leaving the site complies with CWA standards without additional treatment. Among other requirements, SMCRA and OSMRE regulations stipulate that mine operators:

- minimize disturbances and adverse impacts on fish, wildlife, habitat, and hydrologic balance;
- recover the approximate original contour 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).

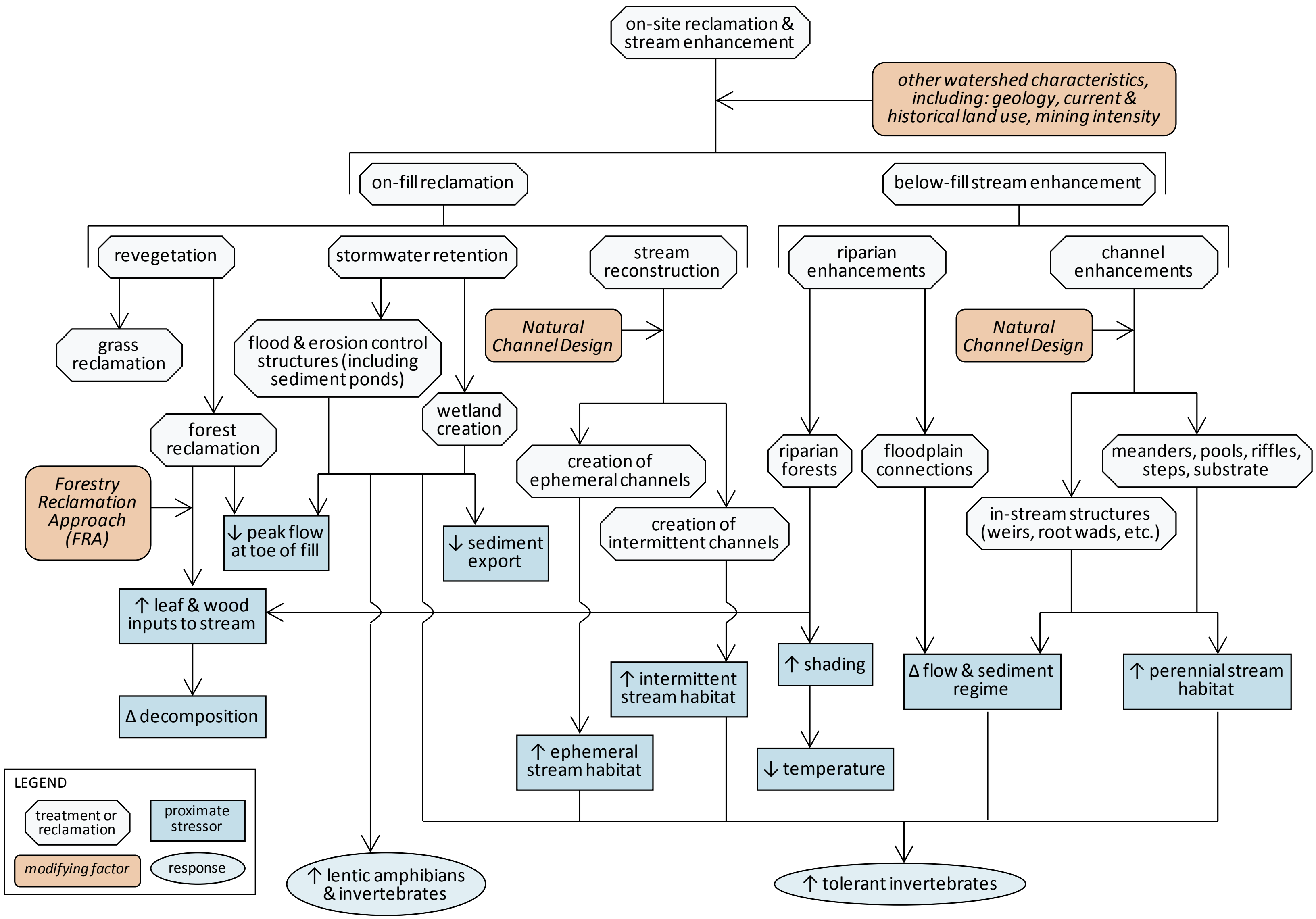

Figure 21. Observed and expected effects of on-site reclamation and stream mitigation efforts. See Section 7 for more discussion and evidence.

To insure the implementation of an approved reclamation plan, activation of a mining permit requires posting of reclamation bonds. If the coal operator forfeits its reclamation responsibilities, these bonds provide funds for the government to complete the work (SMCRA, 30 U.S.C. § 1259). Reclamation bonds are released upon inspection in three phases:

- Phase 1: released after completion of backfilling, placement of homogenized or crush-rock topsoils and contour regrading.
- 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).

7.1.2. Reclamation with Grasses and Pasture

Reclamation techniques developed post-SMCRA traditionally focused on regrading, soil compaction, fast-growing herbaceous vegetation, and stabilization. This was done primarily for erosion control. As a result, vegetative cover at most reclaimed mine sites consists of rapidly growing grasses and legumes, serving as a low-cost, low-risk option for reclamation bond release (Loveland et al., 2003). Pasture and hay lands planted to meet the legal requirements of reclamation could be viable while maintained but might collapse when agronomic practices are neglected after bond release (Barnes et al., 2003).

Soil compaction and competition from herbaceous plants slows the reestablishment of forests on reclaimed mine sites (Bradshaw 1997; Skousen et al., 2009). Minesoils are mixtures of soils, debris, and fractured rock overburden that are spread on reclaimed surfaces to support plant growth (U.S. EPA, 2003, 2005). Compaction of minesoils with the use of heavy equipment during valley fill and reclamation is identified as one of the chief factors limiting the establishment, growth, and survival of trees (Skousen et al., 2006, 2009). As a result, establishment of mid- to late-successional trees could take decades (Skousen et al., 2009).

Evidence suggests that the reclamation approach of heavy soil compaction and planting with grasses largely fails to ameliorate either the hydrologic or water quality impacts of MTM-VF. Ferrari et al. (2009) modeled the flood response in the mined watershed of Georges Creek in Western Maryland and found parallels to what would be expected for urban settings 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). 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 efforts post-SMCRA, while providing stabilization, do not eliminate the deleterious effects of degraded water quality associated with effluent from MTM-VF.

Reclamation with grasses can also alter aquatic communities by decreasing shade and organic inputs to the stream. In small forested watersheds, overhanging trees provide organic matter inputs, while simultaneously reducing photosynthesis by autotrophic organisms (Vannote et al. 1980). This dual effect makes organic inputs the primary source of energy flow into the food web of these streams. In a small headwater stream near Louisville, Kentucky, for example, macroinvertebrate communities relied almost exclusively on leaf inputs (Minshall 1967). Excluding litter artificially from the riparian zone changed the food web structure of a mountainous headwater stream in North Carolina (Wallace et al. 1997).

Finally, non-native grasses and legumes are used for reclamation at most sites (U.S. EPA 2003, 2005, Table 3.J-1) and the disturbance of riparian zones has been linked with increased invasion of non-native plant species (Richardson et al. 2007). Non-native riparian species may in turn modify inputs to the stream (Richardson et al. 2007). However, leaf litter from Japanese knotweed (*Fallopia japonica*), an invasive species found in central Appalachia, did not alter macroinvertebrate assemblages and leaf decomposition rates in an Idaho stream (Braatne et al. 2007).

7.1.3. Forestry Reclamation Approach (FRA)

Recognizing the limitations of these reclamation efforts, the FRA is a set of techniques that has emerged from research conducted over the last several decades to promote the regrowth of forests on reclaimed mine lands (Skousen et al., 2009). Forests were the dominant cover in Appalachia prior to mining activity, and the FRA is being increasingly employed. There are considerable upland benefits to reestablishment of native forests, and there is a large body of literature on tree establishment and growth on reclaimed mine sites (e.g., Ashby, 1997; Andrews et al., 1998; Brinks et al., 2011). This report, however, focuses exclusively on the effects on water quantity, quality, and aquatic communities.

As noted previously, the establishment of trees on reclaimed sites requires low-compaction soils. FRA techniques call for the loose placement of spoil to at least a depth of 4 ft for the establishment of trees (Skousen et al., 2009). Taylor et al. (2009a, b) measured the hydrological characteristics of five test cells created on a surface mine on the Cumberland

Plateau (Ecoregion 68, southwestern Appalachians) of Kentucky. Each test cell contained one of three different types of mining spoil consisting of weathered or unweathered sandstone or a mixture of sandstones and shale. The spoil was loosely dumped following specifications outlined in the FRA (Taylor et al., 2009b). They found that loose placement of mine spoils resulted in greater precipitation infiltration rates and low peak discharge rates. These discharge rates were lower than those from a small, steeply sloped, forested catchment (Taylor et al., 2009a, b). Thus, FRA techniques could help mitigate high downstream peak flows and flooding risk in reclaimed watersheds (Taylor et al., 2009b). To our knowledge, there has been no peer-reviewed study of the impacts of the FRA on water quality. Water infiltrating through the loose spoil will potentially still have elevated concentrations of ions and some metals. However, the effects of forest reclamation on water quality need to be empirically tested.

Besides influencing streamflow, forest reclamation in riparian areas provides shade, temperature moderation, and critical organic inputs to aquatic food webs. Further, the planting of trees rather than grasses promotes habitat for stream-side amphibians. For example, a study of reclaimed mine sites replanted with grasses found a general decline in salamander populations with a concomitant increase in reptiles, particularly snakes, compared to intact forests (Wood et al. 2001). This was likely due to the drier conditions of the grasslands (Wood et al. 2001). Given the positive impacts of replanting native forests, we recommend further study regarding the effects of the FRA on water quality and biota.

7.2. MTM-VF MITIGATION EFFORTS

7.2.1. Overview

In addition to reclamation of postmining sites, placing overburden into stream valleys, if deemed allowable, necessitates mitigation plans. 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; where avoidance is not possible, fills must be minimized. In either case, the proposal must not result in significant degradation to waters of the United States. USACE requirements on Section 404 CWA permits strive for no net loss of aquatic functions (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. Fills resulting in minimal impact, 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 valued aquatic resources are at stake (U.S. EPA, 2003, 2005).

Mitigation activities can be conducted on- or off-site of the permit area. On-site mitigation activities are preferred, but off-site projects might be the only option for lost mountaintop ecosystems and are common in MTM-VF permits. Compensatory mitigation plans include a variety of actions, as indicated below:

- Creating channels using natural stream design techniques to replace streams that have been filled.
- Restoring riparian resources (e.g., revegetation, wetland creation, and floodplain connectivity).
- Enhancing or improving existing stream channels (e.g., riffles/pools, dredging, sinuosity, and bank stabilization).
- Improving fisheries habitat (e.g., shading, increasing habitat heterogeneity, and aeration through riffles or other natural means).
- Controlling sediment and pollution (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).
- Reforesting areas adjacent to mining sites.
- Removing stream encroachments (e.g., roads, crossings, ponds, or other fills).

Below, we address several of these on-site mitigation activities. We examine the extent to which created channels on the fill might replace functions of streams lost from MTM-VF and address the potential for natural stream design and wetland creation to reduce or minimize MTM-VF impacts. Lastly, we discuss stream and channel enhancement and riparian improvements below the fill.

7.2.2. On-Fill Mitigation Efforts

7.2.2.1. *Constructed Channels*

As described in Section 2.1, valley fills generally have rock drains and either a central channel or a set of perimeter channels on or along the fill, which then discharge into sediment retention ponds. The combination of ditches, channels, and ponds are designed to convey runoff for large (e.g., 100-year) storm events. Published studies are generally lacking on whether these ditches and channels can adequately replace the hydrology of natural streams buried by valley fill. Despite this lack of study, some conclusions can be drawn from studies of intact, forested catchments. It should be recognized that stream restoration techniques were developed to restore one or more features of an existing stream with its basic structure intact, not to create streams starting from scratch (Palmer, 2009). Creation of intermittent and perennial streams on filled areas is difficult, in general, due to the inability to capture sufficient groundwater flows to provide a source of streamflow. Lastly, there is no evidence that these channels improve the water quality impacts of MTM-VF or provide habitat invertebrate communities in intermittent or perennial reaches.

The hydrology of a small-order stream is heavily dependent on its connections with its riparian areas, surrounding uplands, and, in perennial or intermittent streams, groundwater. When it rains on an intact, forested watershed, water infiltrates into the soil and moves vertically and laterally depending on subsurface strata (see Figure 5). Further, natural forest soils typically have extensive networks of macropores (spaces in the soil larger than 0.05 mm in diameter), created by old root channels, earthworm and animal burrowing, and freeze-thaw events. These macropores are sites of preferential water flow through the soil, important for the movement of water to the stream channel (Tsuboyama et al., 1994). Steep hillslopes convey water by these shallow subsurface flow paths to the stream (McGuire et al., 2005; McGuire and McDonnell, 2010). In areas of porous bedrock, local groundwater inputs to the stream are also important (Winter, 2007). These natural linkages from the surrounding landscape influence the stream by providing not only stormflow but baseflow during dry periods. The hydrologic flow paths and the amount of time in which the water moves through these flow paths (i.e., residence time) also significantly alter the chemistry of the water entering the stream.

When a mountain is leveled and a valley filled, the hillslopes, subsurface flows, and groundwater exchanges that supported its small streams are permanently dismantled. The degree to which a buried stream was an expression of these connections is likely to determine whether its hydrology can be replaced by a recreated channel. Created channels on valley fills are likely ephemeral or intermittent because they are elevated above the local water table and do not receive significant, year-round groundwater inputs. During a storm event, the created channels might act similarly to an ephemeral stream by conveying water and materials. Where a headwater stream had shallow subsurface and groundwater connections, the hydrology of that stream cannot be replaced. For example, the lost hydrologic function of a perennial stream receiving year-round groundwater inputs cannot be replaced by an ephemeral or intermittent channel. Replacing a perennial stream with an ephemeral channel would cause a shift away from continuous streamflow to one punctuated only by stormflows.

Besides their hydrologic effects, constructed channels and diverted surface flows on valley fills fail to restore water quality and the biological diversity of natural headwaters, particularly in intermittent and perennial reaches. Fritz et al. (2010) found that intermittent and ephemeral constructed channels did not reduce the conductivity of water sampled at the base of the valley fill when compared to forested, reference streams. Further, though peer-reviewed evidence is limited, the available data show that biological assemblages colonizing reclaimed surface waters differ from those of natural headwaters in predictable ways: loss of headwater-specific taxa; increase in tolerant taxa; and presence of taxa adapted to ponds or turbid, slow-moving water (Kirk, 1999; Green et al., 2000; U.S. EPA, 2003, 2005). Total invertebrate density observed on leaf packets was significantly higher in forested streams versus constructed channels or streams at the base of the valley fill (Fritz et al., 2010). Similarly, total taxa richness, EPT richness, and litter decomposition rates (a measure of stream function), were generally higher in forested, perennial, and intermittent reaches compared to perennial streams at the base of the valley fill or intermittent constructed channels on the fill (Fritz et al., 2010). In contrast, decomposition rates and leaf-packet invertebrate diversity did not differ between ephemeral forested streams or ephemeral channels (Fritz et al., 2010).

In addition to invertebrates, salamanders can be useful indicators of water quality due to their small home ranges and relatively stable populations when undisturbed (Welsh and Ollivier, 1998). Many stream salamanders require ephemeral and intermittent streams in forested habitats to maintain viable populations (see Section 3.3). In one study conducted in southern West Virginia, streamside salamander abundance was found to be significantly higher in reference streams compared to those below valley fills (Williams and Wood, 2004). Moreover, using leaf litter bags, 20 salamander larvae were captured in the reference streams versus only 3 salamander larvae in the valley fill streams (see Section 6.1.3) (Williams and Wood, 2004). In contrast to the overall findings of the study, one restored reach supported a large salamander population, possibly due to a strong positive correlation with the numbers of rocks in the stream (Williams and Wood, 2004).

7.2.2.2. *Natural Channel Design*

Creating channels using natural channel design techniques is one mitigation activity that is currently being investigated for the purposes of reestablishing streams on mined lands (Harman et al., 2004). Natural channel design attempts to use the characteristics found in undisturbed streams to promote channel stability and habitat for aquatic organisms. For example, steep headwater streams typically exhibit vertical drop and scour-pool features (Rosgen, 1994). Constructing similar stream-bed morphologies in created channels might more closely mimic natural surface flows.

One of the main goals of natural channel design is to balance sediment export and accumulation, preventing excessive erosion and rapid channel migration (Harman et al., 2004). Given this goal, it may be reasonable to expect natural channel design to reduce sediment export. However, there is no evidence to suggest a mechanism by which natural channel design might significantly reduce elevated conductivity, selenium, or metal concentrations. Absent an improvement in water quality, the enhancement of stream habitat through natural stream design might have little impact on stream biota (see Section 7.2.3.1).

7.2.2.3. *Erosion Control Structures and Constructed Wetlands*

Both erosion control structures and constructed wetlands can reduce sediment runoff, stormflows, and improve water quality. Erosion control structures include riprap or rock-lined channels and sediment ponds. Sediment ponds can improve water quality by removing sediments, suspended solids, and metals (U.S. EPA, 2003, 2005; for a discussion of the effects of ponds on sediment loads, see Section 4.4). Despite this potential, water chemistry data indicate that these ponds fail to prevent downstream water quality degradation (see Section 4), given that sediment ponds are generally present on MTM-VFs as currently constructed.

Constructed wetlands also have the potential to improve water quality. Wetlands can store water, reducing peak stormflows—though this can vary by wetland type (Bullock and Acreman, 2003). Constructed wetlands have also been found to reduce heavy metals, excess nutrients, and total suspended solids (Scholes et al., 1998; Hench et al., 2003; Sheoran and Sheoran, 2006; Vymazal, 2007). Heavy metal removal via wetlands has been shown to be effective in acid mine drainage treatment (Sheoran and Sheoran, 2006). However, constructed wetlands in a waste-water treatment facility failed to reduce conductivity values, and wetlands might become less effective in removing materials over time (Hench et al., 2003). In a study of wetlands on mined lands, almost all the constructed wetlands reduced sediments, while a smaller number provided additional water quality and wildlife benefits (U.S. EPA, 2003, 2005). Wildlife found in wetlands and sediment ponds are generally lentic organisms, not those typically found in Appalachian headwater streams (U.S. EPA, 2003, 2005).

7.2.3. Below-Fill Mitigation Efforts

7.2.3.1. *Riparian Restoration and Stream Channel Enhancement*

Besides mitigation efforts on the valley fill itself, mitigation activities also target the stream reach issuing from below the valley fill. These activities include riparian restoration and stream channel enhancement. Riparian restoration can consist of planting of riparian forests or reconnecting the stream to its floodplain. Stream channel enhancement includes natural channel design techniques of adding structural heterogeneity (e.g., adding boulders and logs, creating riffles and pools) or reforming stream meanders. The planting of riparian forests increases bank stability and reduces erosion, while also adding shade and reducing high temperature extremes. Unconfined stream banks that are devoid of vegetation are more susceptible to channel widening and erosion (Naiman and Decamps, 1997), and woody debris or other instream structures can dissipate water energy and store sediments. Further, riparian zone inputs of leaf litter and wood are critical to aquatic food webs, particularly in headwater catchments (see Section 3) (Tank et al., 2010), and provide critical habitat for amphibians (Wood et al. 2001).

To our knowledge, there is no peer-reviewed, published literature addressing whether riparian restoration or stream enhancement improve water quality and biological assemblages after MTM-VF. These approaches are likely to increase organic matter retention and reduce sedimentation and erosion—but not alleviate elevated ion, selenium or metal concentrations. Therefore, they are likely to have a limited effect on the recovery of instream biota. Though the disturbance of urbanization differs from MTM-VF, many studies in urban settings found that adding structural heterogeneity and restoring riparian areas had limited impact on stream biota when water quality remained degraded (Northington and Hershey, 2006; Sudduth and Meyer, 2006; Bernhardt and Palmer, 2007). The placement of boulders in urban channels has been shown to increase residence times of organic matter (Lepori et al. 2005). Harrison et al. (2004), however, concluded that restoring structural heterogeneity failed to effectively restore macroinvertebrate communities, likely due to the over-riding influence of poor water quality. In a similar fashion, in an acid mine drainage study, water chemistry limited the recovery of benthic organisms beyond any effect of chemical precipitates on the stream substrata (DeNicola and Stapleton, 2002). Biotic assemblages might be able to recover from abrupt downstream disturbances relatively quickly (i.e., a few months to a few years, Wallace, 1990) if connectivity to undisturbed reaches is maintained (Detenbeck et al., 1990; Niemi et al. 1990; Blakely et al. 2006). However, the recovery of stream biota will be limited in cases where poor water quality persists (Niemi et al. 1990).This literature suggests that successful MTM-VF mitigation approaches will need to address water quality before other restoration efforts have a substantial effect.

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. This review focuses on the effects of mountaintop removal operations, which use explosives to remove all or portions of the entire mountain or ridge top, 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 22).

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 include the impacts of these latter 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 springs and ephemeral, intermittent, and small perennial streams that comprise the headwaters of rivers. 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. The excess overburden is disposed of in constructed fills in small valleys or hollows adjacent to the mountaintop site. 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. Coldwater, permanent headwater reaches can be spawning, and nursery areas for native brook trout (*Salvelinus fontinalis,* and headwater reaches are also primary habitat for other native fish such as the creek chub (*Semotilus atromaculatus*), blackside dace (*Phoxinus cumberlandensis*), southern redbelly dace (*Phoxinus erythrogaster*), arrow darter (*Etheostoma sagitta*), and orangethroat darter (*Etheostoma spectabile*; Meyer et al., 2007). 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 also provide habitat for diverse

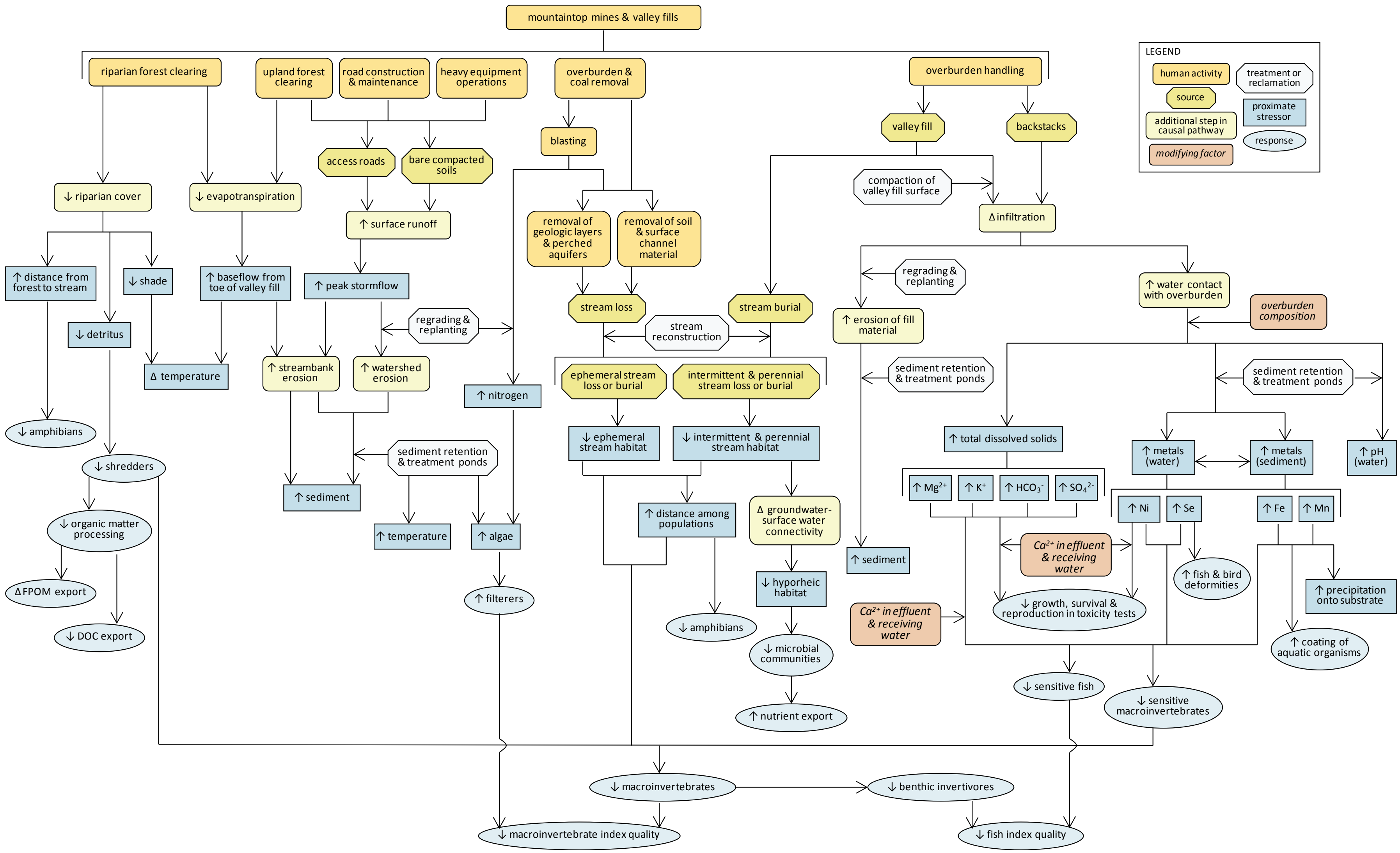

Figure 22. Observed and expected effects of MTM-VF on aquatic ecosystems (narrative description in Section 8.1). (Figure formatted for printing on 11” by 17” paper.)

assemblages of aquatic insects, some of which are unique to these headwater reaches. When a headwater stream is buried under a valley fill, most of the organisms that inhabit it are eliminated. As multiple streams in a mountain range are buried, the distance between the headwaters that remain becomes greater, hindering the movement of biota that is required to sustain populations. 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. Export of nitrogen downstream is expected to increase.

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 might produce greater stormflows, if compaction of the fill surface and mined area by heavy equipment reduces 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 appear to be effective at settling out fine sediments; habitat measures were not strongly related to macroinvertebrate responses. 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 dissolved ions.

Discharges with high concentrations of ions reduced survival of the standard toxicity tests using *C. dubia*. Concentrations of selenium in some streams have been measured at levels that have been shown to cause fish and bird deformities 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 (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 resources
(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 on-site mining reclamation and mitigation activities

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. Our conclusions are consistent with those presented in another recent review of the ecological effects of MTM-VF (i.e., Palmer et al., 2010).

8.2.1. Loss of Headwater 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 literature on headwater stream hydrology and ecology suggests that, while significant, these numbers are likely to underestimate the true magnitude and extent of impairment to regional biodiversity and ecosystem function caused by the loss and burial of headwater catchments.

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 stormflows 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 but not all 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 in a study that observed reduced survival of *C. dubia* 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 *C. dubia*. Tests using the mayfly *Isonychia bicolor* and the amphipod *Hyalella azteca* reported effects on survival at concentrations 3−4 times lower than those affecting *C. dubia*. 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^{2+}, HCO_3^-, and Mg^{2+}. High concentrations of Ca^{2+} and overall hardness might be a mitigating factor. If, as expected, 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 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 Se bioavailability of selenium is difficult to predict, measurements from fish tissue indicated that the Se is elevated in a form that is bioavailable. The greatest risks from exposure and effects would be expected in downstream ponds and reservoirs where Se retention is high and food webs are long.

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 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 mayflieswere most strongly correlated with increased concentrations of ions and selenium and were observed at ion concentrations well below those associated with increased mortality of standard laboratory organisms in short-term tests using mine effluent.

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. Johnson et al. (2010) found that the cumulative level of conductivity values could be predicted using a simple function that combined tributary concentrations with watershed area. This suggests that conductivity levels will accumulate unless mixed with a more dilute source of water—for example, and umined tributary. Pond et al. (2008) showed strong relationships between macroinvertebrate assemblages and conductivity. However, neither Pond et al. (2008) nor Fulk et al. (2003) found additional decreases in fish or macroinvertebrate multimetric indices with greater than one upstream mine or valley fill upstream.

8.2.6. Effectiveness of Mining Reclamation and Mitigation Efforts

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 in Sections 4 and 6 were observed after sites were reclaimed for 3 to 24 years, the effects would be expected to persist. Preliminary results suggest that FRA techniques allow increased infiltration and smaller peak stormflows, possibly decreasing the risk of downstream flooding. Further study on water quality impacts of this reclamation approach is needed.

Channels created on valley fills are no longer associated with the same geologic structures and hydrologic flow paths that preceded MTM-VF. Stream restoration techniques were developed to restore one or more features of an existing stream, not to create entirely new streams. During precipitation events, created channels might act similarly to an ephemeral stream by conveying water and materials, but the hydrology of a buried perennial or intermittent reach is unlikely to be replaced. There is no evidence that these channels, regardless of permanence of flow, improve water quality, and invertebrate diversity and decomposition rates are generally lower in constructed channels or at the base of the fill compared to forested perennial or intermittent streams. In contrast, invertebrate diversity might not differ in constructed channels and ephemeral streams. Increasing stream structural heterogeneity and restoring riparian areas below the fill are likely to have a limited effect on aquatic biota, given the failure to address the underlying problem of water quality.

8.3. INFORMATION GAPS, ASSESSMENT NEEDS 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 with 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 mountaintops 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. 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.

To support quantification of cumulative effects (see Section 8.3.7), it could be appropriate to extend these surveys beyond the linear stream miles directly altered by mountaintop mining and buried by valley fills to consider the fractal nature of these stream networks at larger scales. This would involve quantifying the proportional area of individual larger watersheds of a certain size affected by mountaintop mines and valley fills. If sufficient comprehensive remote sensing or aerial photo interpretation is or becomes available, the landscape alterations by MTM-VF might be measured in terms of the area and volume of earth movement, the change in vegetation cover and type, and the proximity of these activities to stream channels.

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 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 might 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

Declines in macroinvertebrate indices were observed at ion concentrations well below those associated with increased mortality of standard laboratory test organisms in short-term tests, indicating that some native organisms are more sensitive to the constituents in mining effluents. 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. Increasing understanding of toxic mechanisms could help interpret the responses observed from biological surveys. 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 *C. 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, and conductivity 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 and conductivity, and slightly alkaline pH. These waters also have relatively high concentrations of individual ions, such as SO_4^{2-}, HCO_3^-, Ca^{2+}, and Mg^{2+}. 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, *C. dubia* and *D. magna*, and the amphipod, *H. 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 *D. 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 *C. tentans* and *I. bicolor*, might not be good surrogates. Aquatic *Diptera* possess anal papillae, which though different in structure are functionally equivalent to chloride epithelia. In the case of *I. bicolor*, its natural distribution is in larger streams with greater conductivities than those found in the streams affected by MTM-VF (Kondratieff and Voshell, 1984). Also, the bioassays testing *I. 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. Conduct Mesocosm and Microcosm Experiments with Indigenous Taxa

An alternative approach to the single species bioassays would be the use of mesocosm and microcosm experiments to further investigate the causal relationships between elevated conductivity and the ions associated with MTM-VF and benthic macroinvertebrate assemblages. Such an approach has been described by Clements (2004), who investigated metals associated with mine drainage from hardrock metal mining. As described by Clements (2004), plastic trays filled with substrates from the streambed were placed in an unimpaired stream for 40 days to allow colonization by the indigenous benthic macroinvertebrate assemblage. In the mesocosm experiment, the colonized trays were placed in a stream impaired by the contaminants of interest. In the microcosm experiment, the colonized trays were placed in artificial streams dosed with one or mixtures of the contaminants of interest. After a 10-day exposure period, the trays were removed and the macroinvertebrates processed for identification and counting. The results from the mesocosm experiment verify that the effects observed in field samples occur as a result of exposure to the contaminants of interest, whereas the results of the microcosm would quantify the levels of the contaminants of interest that cause those effects.

8.3.6. 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 could, 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 could be somewhat elevated in effluent waters. Therefore, data on sediment concentrations of metals could be used to assess whether sediment contamination is 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.

8.3.7. Quantify Cumulative Effects

Cumulative impact assessment can be approached from many perspectives. The impacts of MTM-VF can be evaluated as more land is mined over time. Impacts can be approached from a watershed perspective, evaluating the overall impact of all human activities that influence aquatic ecosystems. Or, the influence of stressors from MTM-VF can be traced through different components of the food web, evaluating effects on ecosystem functions or movement through the food web. Another perspective might evaluate impacts of upstream alterations from MTM-VF on downstream ecosystem functions. Among other species, long-lived unionid mussels might be used to monitor effects on ecosystem functions and food webs.

We found little published literature that evaluated cumulative impacts from any of these perspectives in our study region. Johnson et al. (2010) successfully modeled changes in conductivity levels as tributaries combine but did not link these changes to biological endpoints. Petty et al. (2010) evaluated the impact of increasing mining intensity in watersheds influenced primarily by acid mine drainage. Impacts increased with intensity of mining but were also influenced by underlying coal geology, and the spatial configuration of disturbance. Several papers have documented that exposure to one stressor can make ecological systems more vulnerable to the impacts from subsequent stressors or disturbances, potentially preventing recovery (e.g., Brooks et al., 2007; Clements et al., 2008; Paine et al., 1998). However, none of the studies have investigated the stressors or aquatic ecosystems associated with MTM-VF. The movement of selenium through ecosystems has been reviewed (Chapman et al., 2010), but no studies were identified from the aquatic ecosystems of the central Appalachian coalfields.

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. Concurrent samples of the biological assemblages could be used to develop models to predict the temporal and spatial extent of impacts. 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 extensive 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, might be needed.

8.3.8. Quantify Longitudinal Effects

With the exception of Johnson et al. (2010), which only measured conductivity longitudinally in a single 118-km^2 catchment, no studies have quantified the change in conductivity, individual ion concentrations, pH, or precipitates as the water progresses downstream in a catchment with valley fills. This is needed to quantify the potential longitudinal effects of these elevated ions downstream from valley fills and understand how the relative concentrations of the ions change over space. Moreover, if a catchment has more than one valley fill, as did Johnson et al. (2010), such studies will contribute to our understanding of cumulative effects on downstream water chemistry. This could also include sampling upstream, in, and downstream of the sedimentation ponds to further quantify any separate effects of the sedimentation ponds.

8.3.9. Quantify Effects on Stream Hyporheic Zones

No studies have investigated the effects of MTM-VF on the hyporheic zones of streams downstream from valley fills or even other groundwater resources. To fully understand the longitudinal and cumulative effects of MTM-VF, such data are needed. Questions include, is the chemistry of interstitial waters similar to that of the surface water and how might interaction with groundwaters affect this chemistry? Are there similar adverse effects to invertebrates and other organisms living in downstream hyporheic zones?

8.3.10. Quantify Functionality of Constructed Streams and Mitigation Efforts

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.

We limited our review of reclamation and mitigation activities to evidence of their effects on on-site water quality, quantity, and aquatic ecosystems. Many research and development needs remain: methods for decreasing concentrations of ions and improving water quality; research on the long-term downstream impacts from disturbance, burial and loss of headwater streams, including physical impacts on sediment supply, hydrology, and geomorphology and their implications for stream stability and channel adjustment during and post-mining; and research on improving stream channels enhancements for areas that have no reference streamflow curves or gauged streams.

Off-site mitigation of a wide variety of stream impacts, for example, from agriculture or development, can be used to offset impacts from MTM-VF. The quantification of the effectiveness of these efforts and the degree to which they compensate for the losses from MTM-VF is an area of active work and would form the basis of a useful review.

8.3.11. Expand the Scope of Review to Include Evidence from Non-Peer-Reviewed Sources and Terrestrial Impacts

We limited our source material for the current report to the published peer-reviewed literature. Evidence reported in theses, dissertations, non-peer-reviewed symposia and reports, could, after equivalent documentation and quality review of methods and analyses, contribute additional insights.

The scope of our report is limited to impacts on aquatic ecosystems. An assessment of impacts of MTM-VF on terrestrial ecosystems would provide a useful companion document.

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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^{SM} 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. Additional sources were suggested by the Science Advisory Board and public commenters. In total, over 500 sources of information were identified, including books, conference proceedings, journal articles, government reports, and theses and dissertations.

All potential sources were evaluated for peer-review status. Suggested conference proceedings, in general, did not meet this criterion, except for the Proceedings of the American Society of Mining and Reclamation, which has used a review process since 2002 (Richard Barnhisel, personal communication), and the 41st Symposium of the British Ecological Society, Ecology: Achievement and Challenge (Lindsay Haddon, personal communication). Peer-reviewed sources were classified by region and relevance (see Table A-1). The region of interest was defined as the central Appalachian coalfields (see Figure 1). Laboratory studies were included in the category as "stressors in streams from other regions." Most of the sources judged to be not relevant focused on acid mine drainage, rather than the alkaline discharges that are typical of mountaintop mines and valley fills.

Table A-1. Categorization of literature retrievals by region and relevance

Description	Number of citations
MTM-VF in region of interest	18
MTM-VF in other region	0
Stressors in streams of interest	24
Stressors in streams from other regions	83
Review article of stressor of interest	51

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-2). The search covered publication dates up to December 2010. 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 are listed first.

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.

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 ($CaSO_4$), $MgSO_4$, 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 the citations identified, 193 were considered to be applicable and relevant to organism groups of interest (see Table A-3). 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-2. Keywords used for ISI Web of Knowledge℠ and Google™ Scholar searches

Keywords			
Algae	DO	Mg	sediment transport
Alkaline	DO	mine reclamation	Sediments
Amphibian	electrical conductivity	Minnow	Selenium
Anuran	Ephemeroptera	Mollusk	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

DO = dissolved oxygen; Mg = magnesium.

Table A-3. 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 Reclamation and Enforcement (OSMRE) and delegated states with OSMRE 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.[15]

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,

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[15]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) and 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).

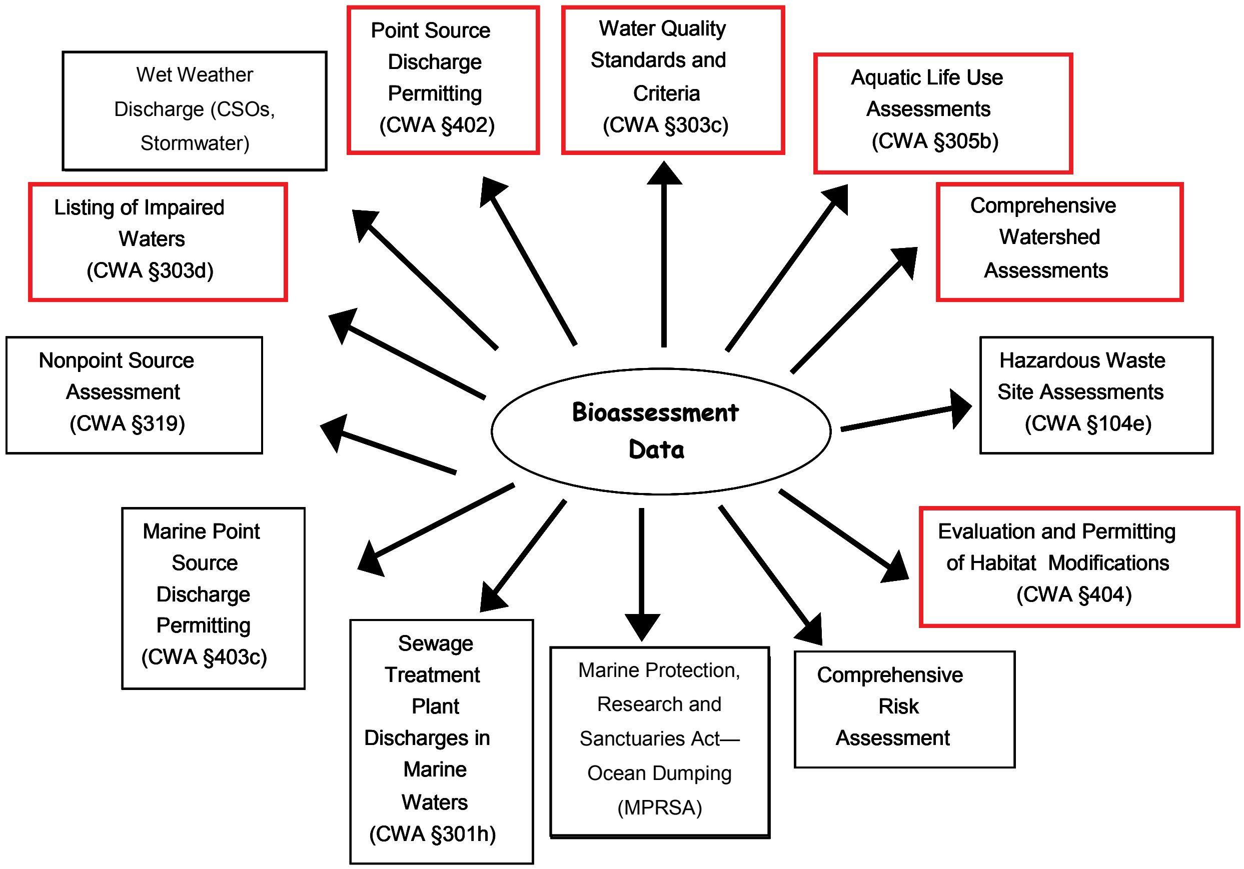

Figure B-1. Simple representation of CWA programs that rely on biological assessment data for program implementation. Coal mining activities sections highlighted in red.

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 in its biological assessment program. 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 (WVSCI), 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 WVSCI 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 by-products 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, molluscs 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.

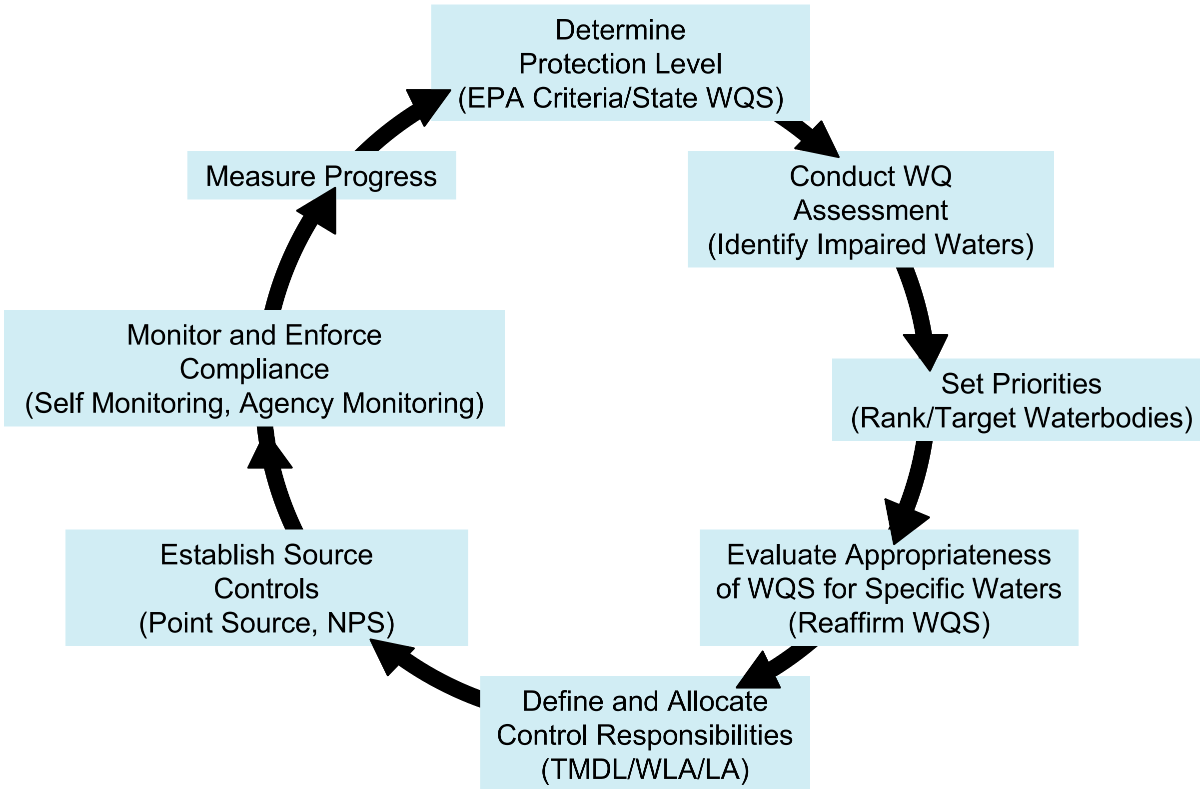

Figure B-2. Water quality-based approach to pollution control.