Effects of Abandoned Coal-Mine Drainage on Streamflow and Water Quality in the Mahanoy Creek Basin, Schuylkill, Columbia, and Northumberland Counties, Pennsylvania, 2001

by Charles A. Cravotta III

In cooperation with the Schuylkill Conservation District and
the Pennsylvania Department of Environmental Protection

Scientific Investigations Report 2004-5291

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
Gale A. Norton, Secretary

U.S. Geological Survey
Charles G. Groat, Director

U.S. Geological Survey, Reston, Virginia: 2005
For sale by U.S. Geological Survey, Information Services
Box 25286, Denver Federal Center
Denver, CO 80225

For more information about the USGS and its products:
Telephone: 1-888-ASK-USGS
World Wide Web: http://www.usgs.gov/

Suggested citation:
Cravotta, C.A., III, 2005, Effects of abandoned coal-mine drainage on streamflow and water quality in the Mahanoy Creek Basin, Schuylkill, Columbia, and Northumberland Counties, Pennsylvania, 2001: U.S. Geological Survey Scientific Investigations Report 2004-5291, 60 pages plus appendixes.

Library of Congress Cataloging-in-Publication Data

Contents

Figures

Tables

Appendixes

Conversion Factors and Datum

Multiply	By	To obtain
Length		
inch (in.)	25,400	micrometer (µm)
inch (in.)	2.54	centimeter (cm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
Area		
acre	4,047	square meter (m^2)
square mile (mi^2)	2.590	square kilometer (km^2)
Flow rate or volume		
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
cubic foot per second (ft^3/s)	1,699	liter per minute (L/min)
gallon per minute (gal/min)	3.785	liter per minute (L/min)
gallon (gal)	3.785	liter (L)
Mass		
ton, short (2,000 lb)	0.9072	megagram (Mg)
ounce	0.035	gram (g)
Application or loading rate		
pounds per year (lb/yr)	0.4536	kilograms per year (kg/yr)
pounds per acre per day [(lb/acre)/d]	0.1121	grams per square meter per day [(g/m^2)/d]
pounds per acre per day [(lb/acre)/d]	1.121	kilograms per hectare per day [(kg/ha)/d]

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

°F = (1.8 x °C) + 32

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

°C = (°F - 32) / 1.8

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L). Concentrations of constituents in sediment are given as micrograms per gram (µg/g) or weight percent (wt%).

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25°C).

Symbols are used to indicate less than (<), less than or equal to (≤), greater than (>), greater than or equal to (≥), equal to (=), and approximately equal to (~).

Effects of Abandoned Coal-Mine Drainage on Streamflow and Water Quality in the Mahanoy Creek Basin, Schuylkill, Columbia, and Northumberland Counties, Pennsylvania, 2001

by Charles A. Cravotta III

Abstract

This report assesses the contaminant loading, effects to receiving streams, and possible remedial alternatives for abandoned mine drainage (AMD) within the Mahanoy Creek Basin in east-central Pennsylvania. The Mahanoy Creek Basin encompasses an area of 157 square miles (407 square kilometers) including approximately 42 square miles (109 square kilometers) underlain by the Western Middle Anthracite Field. As a result of more than 150 years of anthracite mining in the basin, ground water, surface water, and streambed sediments have been adversely affected. Leakage from streams to underground mines and elevated concentrations (above background levels) of acidity, metals, and sulfate in the AMD from flooded underground mines and (or) unreclaimed culm (waste rock) degrade the aquatic ecosystem and impair uses of the main stem of Mahanoy Creek from its headwaters to its mouth on the Susquehanna River. Various tributaries also are affected, including North Mahanoy Creek, Waste House Run, Shenandoah Creek, Zerbe Run, and two unnamed tributaries locally called Big Mine Run and Big Run. The Little Mahanoy Creek and Schwaben Creek are the only major tributaries not affected by mining. To assess the current hydrological and chemical characteristics of the AMD and its effect on receiving streams, and to identify possible remedial alternatives, the U.S. Geological Survey began a study in 2001, in cooperation with the Pennsylvania Department of Environmental Protection and the Schuylkill Conservation District.

Aquatic ecological surveys were conducted by the U.S. Geological Survey at five stream sites during low base-flow conditions in October 2001. Twenty species of fish were identified in Schwaben Creek near Red Cross, which drains an unmined area of 22.7 square miles (58.8 square kilometers) in the lower part of the Mahanoy Creek Basin. In contrast, 14 species of fish were identified in Mahanoy Creek near its mouth at Kneass, below Schwaben Creek. The diversity and abundance of fish species in Mahanoy Creek decreased progressively upstream from 13 species at Gowen City to only 2 species each at Ashland and Girardville. White sucker (*Catostomus commersoni*), a pollution-tolerant species, was present at each of the surveyed reaches. The presence of fish at Girardville was unexpected because of the poor water quality and iron-encrusted streambed at this location. Generally, macroinvertebrate diversity and abundance at these sites were diminished compared to Schwaben Creek and other tributaries draining unmined basins, consistent with the observed quality of streamwater and streambed sediment.

Data on the flow rate and chemistry for 35 AMD sources and 31stream sites throughout the Mahanoy Creek Basin were collected by the USGS during high base-flow conditions in March 2001 and low base-flow conditions in August 2001. A majority of the base-flow streamwater samples met water-quality standards for pH (6.0 to 9.0); however, few samples downstream from AMD sources met criteria for acidity less than alkalinity (net alkalinity = 20 milligrams per liter as $CaCO_3$) and concentrations of dissolved iron (0.3 milligram per liter) and total manganese (1.0 milligram per liter). Iron, aluminum, and various trace elements including cobalt, copper, lead, nickel, and zinc, were present in many streamwater samples at concentrations at which continuous exposure can not be tolerated by aquatic organisms without an unacceptable effect. Furthermore, concentrations of sulfate, iron, manganese, aluminum, and (or) beryllium in some samples exceeded drinking-water standards. Other trace elements, including antimony, arsenic, barium, cadmium, chromium, selenium, silver, and thallium, did not exceed water-quality criteria for protection of aquatic organisms or human health. Nevertheless, when considered together, concentrations of iron, manganese, arsenic, cadmium, chromium, copper, lead, nickel, and zinc in a majority of the streambed sediment samples from Mahanoy Creek and AMD-affected tributaries exceeded the probable effect level for toxicity because of multiple contaminants.

The water-quality data for the AMD sources were used to determine priority ranks of the sources on the basis of loadings of dissolved metals (iron, manganese, and aluminum), net alkalinity, and sulfate and to identify possible remedial alternatives, including passive-treatment options. The ranking sequence for the top AMD sources based on the high base-flow data generally matched that based on the low base-flow data. Although concentrations increased with decreased flow, the pollutant loadings generally increased with flow; six previously identified intermittent AMD sources were not discharging during the low base-flow sampling period. The top 4 AMD sources, Locust Gap Tunnel (M29), Packer #5 Breach (M13), Packer #5 Borehole (M12), and Girard Mine seepage (M11), on the basis of dissolved metals loading in March 2001, accounted for more than 50 percent of the metals loading to Mahanoy Creek, whereas the top 15 AMD sources accounted for more than 99 percent of the metals loading. When sampled in March 2001, the top 15 AMD sources had flow rates ranging from 0.4 to 17.2 cubic feet per second (680 to 29,200 liters per minute) and pH from 3.9 to 6.7. Dissolved iron was the principal source of acidity and metals loading; concentrations of iron ranged from 2.1 to 18 milligrams per liter. Dissolved manganese ranged from 0.95 to 6.4 milligrams per liter. Dissolved aluminum exceeded 1.0 milligram per liter at 4 of the top 15 AMD sources but was less than 0.4 milligram per liter at the others. Nine of the top 15 AMD sources, including the top 4, were net alkaline (alkalinity greater than acidity); the others were net acidic and will require additional alkalinity to facilitate metals removal and maintain near-neutral pH.

Alkalinity can be acquired by the dissolution of limestone and (or) bacterial sulfate reduction within various passive-treatment systems including anoxic or oxic limestone drains, limestone-lined channels, or compost wetlands. Subsequently, the gradual oxidation and consequent precipitation of iron and manganese can be accommodated within settling ponds or aerobic wetlands. Assuming an iron removal rate of 180 pounds per acre per day (20 grams per square meter per day), constructed treatment wetlands at the top 15 AMD sites would require a minimum area ranging from 0.3 to 5.8 acres (1,210 to 23,500 square meters). Land area below the Packer #5 Breach (M13, ranked 2nd of all AMD sources in the watershed), the Packer #5 Borehole (M12, ranked 3rd), and the Centralia Tunnel (M19, ranked 6) may be sufficient for installation of passive treatment. However, because of the proximity of the Locust Gap Tunnel and many other discharges to streams, roads, or railroads, and the limited availability or access to land at the discharge location, passive treatment would not be suitable at many AMD sites. The reduction of infiltration and removal of culm waste and (or) the relocation of the discharge to nearby areas could decrease the AMD quantities and facilitate treatment at some of the priority AMD sites.

Introduction

Mahanoy Creek is a tributary of the Susquehanna River that drains an area of 157 mi^2 (407 km^2) in east-central Pennsylvania (fig. 1). The Mahanoy Creek Basin is a narrow watershed bounded by a series of parallel, westward-trending ridges. Mahanoy Creek flows 53.7 mi (86.4 km) from its headwaters near the village of Buck Mountain in Schuylkill County to its mouth near Herndon in Northumberland County. From the village of Buck Mountain, Mahanoy Creek flows westward through Mahanoy City, Girardville, and Ashland. From Ashland, Mahanoy Creek flows southwestward approximately 2 mi (3.2 km) to Gordon and then another 35 mi (56 km) westward to its mouth on the Susquehanna River.

The upper part of Mahanoy Creek, with a total drainage area of 46 mi^2 (119 km^2) at Ashland, drains an area of 37 mi^2 (96 km^2) underlain by the Western Middle Anthracite Field (fig. 1), where anthracite was extensively mined from about 1840 through 1950 (Reed and others, 1987). This area includes all or most of the drainage areas for the upper Mahanoy Creek, North Mahanoy Creek, and Shenandoah Creek plus drainage from the vicinity of Centralia locally named “Big Mine Run” (Sanders & Thomas, Inc., 1975). Smaller areas than described above also have been extensively mined including approximately 2 mi^2 (5.2 km^2) for an unnamed tributary, locally named “Big Run” (Sanders & Thomas, Inc., 1975) that drains from Mahanoy Mountain southward to Mahanoy Creek at Locustdale and 3 mi^2 (7.8 km^2) for the upper part of Zerbe Run in the western part of the basin south of Trevorton (fig. 1). Additionally, an intermediate reach of Mahanoy Creek, near Helfenstein, receives discharges from the Locust Gap (M29) and Doutyville (M31) Tunnels that drain a mined area of 7 mi^2 (18.1 km^2) to the north within the adjoining Shamokin Creek Basin (Reed and others, 1987; Cravotta and Kirby, 2004a).

Leakage from streams to underground mines and elevated concentrations (above background levels) of acidity, metals, and sulfate in the abandoned mine drainage (AMD) from flooded underground mines and (or) unreclaimed culm (waste rock) piles degrade the aquatic ecosystem and impair uses of the main stem of Mahanoy Creek from its headwaters to its mouth on the Susquehanna River (Sanders & Thomas, Inc., 1975; Reed and others, 1987; Pennsylvania Department of Environmental Protection, 2001, 2002, 2004). Degradation also is evident within various tributaries, including North Mahanoy Creek, Waste House Run, Shenandoah Creek, Big Mine Run, Big Run, and Zerbe Run. The Little Mahanoy Creek and Schwaben Creek are the only major tributaries not affected by mining. Consequently, Mahanoy Creek is designated a “high priority watershed” on the Pennsylvania Department of Environmental Protection (PaDEP) degraded watershed list

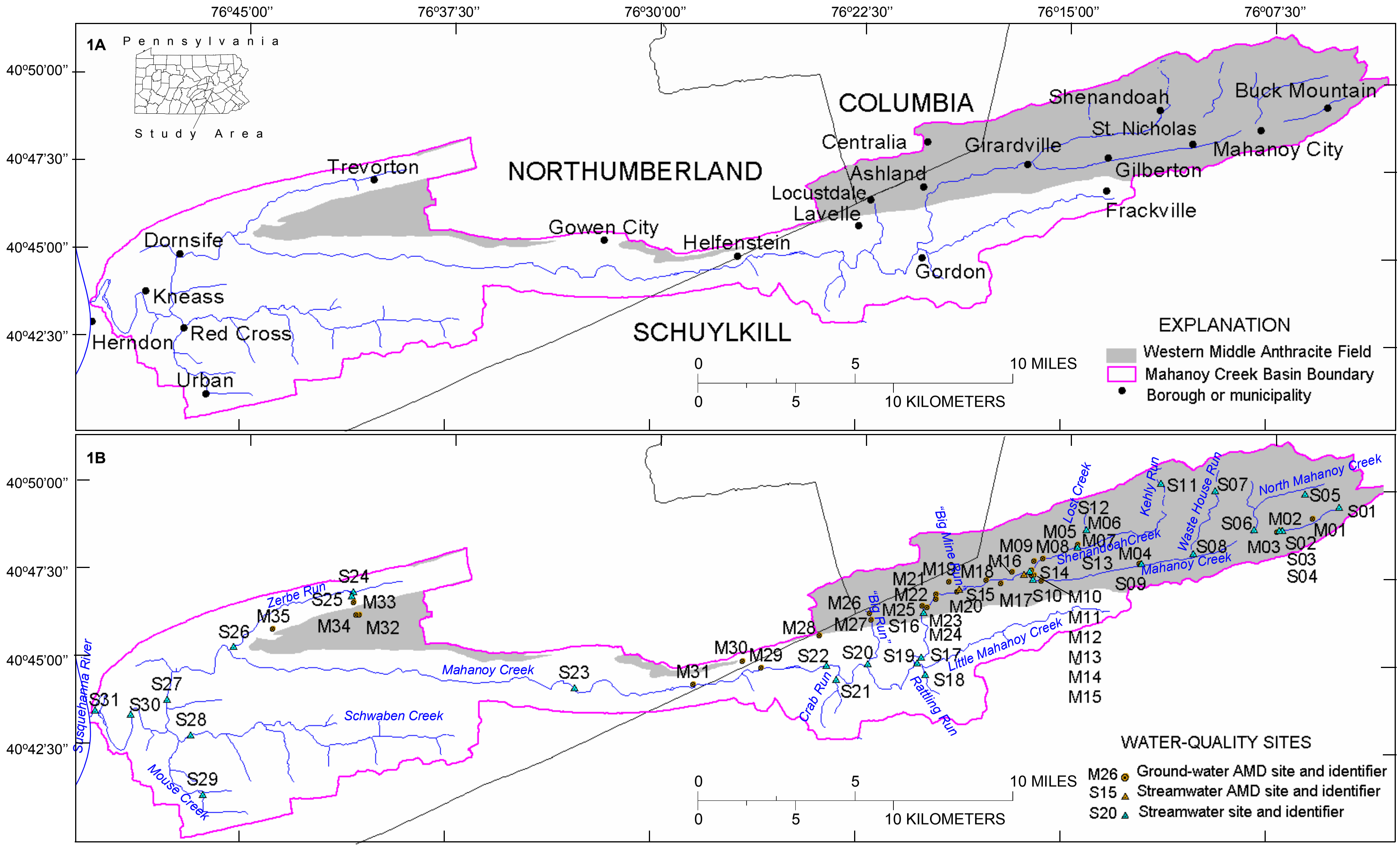

Figure 1. Mahanoy Creek Basin, Pennsylvania, showing locations of (A) municipalities and Western Middle Anthracite Field, (B) named streams and corresponding streamwater and mine-drainage monitoring locations. Mine-drainage sites that discharge directly from ground designated ground-water AMD, and those that were downflow within channel or ditch designated streamwater AMD. Western Middle Anthracite Field corresponds with mapped area of Llewellyn Formation after Berg and others (1980). Hydrography from U.S. Geological Survey 1:100,000 topographic map base.

(Pennsylvania Department of Environmental Protection, 1998, 2004). To assess the current effects of AMD and to identify possible remedial alternatives, the U.S. Geological Survey (USGS) began a study in 2001, in cooperation with the PaDEP and the Schuylkill Conservation District (SCD). The study was initiated in response to a request for assistance by the Mahanoy Creek Watershed Association (MCWA).

Purpose and Scope

This report assesses the contaminant loading, effects to receiving streams, and possible remedial alternatives for AMD within the Mahanoy Creek Basin on the basis of water-quality data collected by the USGS during 2001 and supplemental ecological data collected by the USGS and PaDEP during 2000-2002. Data on the flow rate and chemistry of water were collected by USGS at the known AMD sites and at selected stream sites within the Mahanoy Creek Basin during high base-flow conditions in March 2001 and then repeated during low base-flow conditions in August 2001. During the low base-flow survey in August 2001, streambed sediments were collected for chemical analysis, and during October 2001, data on the diversity and biomass of fish species were collected by USGS at five of the sampled stream sites. Additionally, during 2000-2002, aquatic macroinvertebrate samples were collected by PaDEP throughout the Mahanoy Creek Basin. The basin-wide synoptic monitoring of flow and water quality during stable base-flow conditions was performed to (1) identify site-specific characteristics including temporal variability associated with seasonal changes in base flow, (2) indicate spatial variability and relative effects of the AMD throughout the basin, and (3) avoid complications in data collection and interpretation associated with rainfall or other short-term weather events. The supplemental surveys of streamflow and aquatic ecology documented the effects of the AMD on streamwater resources. The study data were compiled into a digital database and a geographic information system (GIS). The data were used to compute contaminant loading rates, to determine the potential effects of the AMD on aquatic ecology, and to identify possible remedial priorities and alternatives for watershed rehabilitation. The PaDEP, SCD, and MCWA will use study results for selecting AMD priorities and remediation alternatives.

Physiography and Land Use

The Mahanoy Creek Basin lies in the Appalachian Mountains section of the Ridge and Valley Physiographic Province (Way, 1999). This area is characterized by complexly deformed strata and elongate, northeast-southwest trending ridges that bound narrow valleys. Because of their steep slopes and thin rocky soils, the ridges tend to be forested and sparsely developed. Urban, industrial, and agricultural land uses predominate in the valleys. The western two-thirds, or lower part, of the Mahanoy Creek Basin is in the Northern Shale Valleys and Slopes Ecoregion where forested and agricultural land uses predominate (Pennsylvania Department of Environmental Protection, 2001). The eastern third, or upper part, of the basin is in the Anthracite Coal Ecoregion (Pennsylvania Department of Environmental Protection, 2001).

Land use in the Mahanoy Creek Basin has been classified as 66 percent forested, 21 percent agricultural, 9 percent "barren, mined," and 4 percent urban (U.S. Geological Survey, 2000; Pennsylvania Department of Environmental Protection, 2002). Nevertheless, this land-use classification may be misleading because, in the upper basin, abandoned underground mines extend beneath much of the surface, and "natural" reforestation conceals large tracts of unreclaimed spoil. Considering the extent of "spoil" and other land-use patterns depicted by USGS topographic maps, land use in the upper Mahanoy Creek Basin at Ashland could be classified as 50 percent forested, 44 percent "mine spoils," 5 percent urban, and less than 1 percent agricultural. Hence, although the anthracite industry largely is inactive, mining could be considered a major land use within the valley of the upper basin.

Geology and Mining History

The Western Middle Anthracite Field underlies parts of the Mahanoy Creek Basin and the neighboring Shamokin Creek Basin (Reed and others, 1987; Cravotta and Kirby, 2004a). The coalfield is a synclinal basin, or "canoe-shaped" structure, that has been subdivided by parallel faults. Sandstone, siltstone, and conglomerate are the dominant lithologies surrounding the coalbeds; limestone has not been mapped within the coalfield (Wood and others, 1986; Berg and others, 1989; Eggleston and others, 1999). In the Mahanoy Creek Basin, a total of 24 coalbeds of the Llewellyn and Pottsville Formations of Pennsylvanian Age, with average thicknesses from 0.6 to 8.3 ft (0.2 to 2.5 m), have been identified and mined to depths exceeding 2,500 ft (762 m) below land surface (Reed and others, 1987). Most anthracite mines were developed as large underground complexes or "collieries" in the valleys, where shafts and tunnels connected mine drifts and slopes within multiple coalbeds. Generally, mining was conducted by the room-and-pillar method, with barrier pillars left intact between adjacent collieries (Reed and others, 1987).

Anthracite production in Pennsylvania peaked in 1917 at more than 100 million short tons, including 12 million tons from the Mahanoy Creek Basin (Sanders & Thomas, Inc., 1975; Pennsylvania Department of Environmental Protection, 2001; Pennsylvania Coal Association, 2001). During the peak of production, 95 percent of the anthracite was obtained from underground mines. In 2000, less than 6 million short tons of anthracite were produced in Pennsylvania; only 5.5 percent was from underground mines (Pennsylvania Coal Association, 2001). Only a few surface and deep mines presently are active in the Mahanoy Creek Basin; the majority of these operations involve reprocessing of coal waste piles (Pennsylvania Department of Environmental Protection, 2002).

Substantial quantities of coal were placed with other rock on waste piles near the historically active mining and processing facilities because of the vast amounts of coal available and a limited market for fine-grained material. The waste material generally can be characterized as rock (1-5 percent coal), culm (20-80 percent coal), or silt (20-80 percent coal), with culm being coarser than silt (Sanders & Thomas, Inc., 1975). Culm and rock banks are concentrated near the abandoned deep mines and collieries, and much of the silt has eroded from piles and accumulated in the streambeds. According to Sanders & Thomas, Inc. (1975), some of the larger culm and rock banks are near the eastern limits of Girardville, between Gilberton and St. Nicholas, and east of Centralia. Some of the larger silt piles in the watershed are along North Mahanoy Creek, along the southern side of Mahanoy Creek between Mahanoy City and Gilberton, on the north side of Mahanoy Creek in the vicinity of St. Nicholas, near the eastern limits of Centralia, and along Zerbe Run west of Trevorton. The heaviest silt deposits are concentrated along Mahanoy Creek from below Ashland, downstream past Gordon, and along Zerbe Run southwest of Trevorton.

Most underground mines were closed before 1960 and, when ground-water pumping ceased, the mines flooded producing underground “mine pools.” When the underground mines were active, some stream channels were lined artificially or diverted to reduce leakage and to reduce the costs of pumping ground water from the mines. Upon closure of the mines, leakage resumed from sections of these channels, and some streams coursed into open mine pits or other openings. For example, during dry periods, upper Mahanoy Creek, North Mahanoy Creek, Waste House Run, and Lost Creek can lose all or most of their flow to underground mines (Sanders & Thomas, Inc., 1975; Pennsylvania Department of Environmental Protection, 2004).

Numerous areas within the Mahanoy Creek valley and along the valley slopes have not been reclaimed or revegetated. Barren, steep banks of rock spoil, culm, and silt are sources of sediment (suspended solids), acidity, metals, and sulfate in water that infiltrates or runs off the surface. In parts of the basin, surface flow is diverted through subsidence pits, fractures, and mine openings to the underground mines. In downstream reaches, the water resurges as AMD adding contaminants to Mahanoy Creek and its tributaries, while contributing substantially to streamflow. The AMD constitutes a substantial portion of base flow in mined watersheds of the region, especially during low-flow or drought conditions (Becher, 1991).

Discharges typically emanate from tunnels, slopes, air shafts, and other passages, including fractures in stream channels, and other topographically low points overlying the mine complexes. Many of the same streams that lose water to underground mines in the upper reaches gain water from mine discharges in their lower reaches. In some cases, the mine complexes extended beyond surface-water divides, enabling the transfer of surface and ground water between adjacent stream basins. For example, water originating in the Shamokin Creek Basin discharges to the Mahanoy Creek Basin from the Doutyville, Helfenstein, Locust Gap, and Centralia Mine discharges, whereas water is conveyed from the Mahanoy Creek Basin through interconnected mine complexes to become part of the overflow from the Henry Clay Stirling discharge in the Shamokin Creek Basin (Reed and others, 1987; Cravotta and Kirby, 2004a). Because the underground mine complexes were extensive, their discharge volumes tend to be substantially greater and more continuous than those from less extensive surface mines or spoil piles.

Water Quality

Generally, AMD can have low pH and elevated concentrations (above background) of dissolved sulfate (SO_4^{2-}), iron (Fe^{2+}, Fe^{3+}), manganese (Mn^{2+}), aluminum (Al^{3+}), and other metals that result from the oxidation of pyrite (FeS_2) and the dissolution of carbonate, oxide, and aluminosilicate minerals by acidic water (Cravotta, 1991; Rose and Cravotta, 1998). Pyrite oxidation takes place where oxygen (O_2) and moisture (H_2O) are available as described in the reactions below.

$$FeS_2 + 3.5\ O_2 + H_2O \rightarrow Fe^{2+} + 2\ SO_4^{2-} + 2\ H^+ \qquad (1)$$

$$Fe^{2+} + 0.25\ O_2 + H^+ \rightarrow Fe^{3+} + 0.5\ H_2O \qquad (2)$$

$$Fe^{3+} + 3\ H_2O \rightarrow Fe(OH)_3 + 3\ H^+ \qquad (3)$$

Initially, pyritic sulfur is oxidized to SO_4^{2-} (reaction 1). Under non-recharge conditions, various secondary sulfate minerals and concentrated, acidic fluids can accumulate near the pyrite. Subsequently, infiltrating water or surface runoff can dissolve and transport the acidic oxidation products. In contrast with SO_4^{2-}, which is transported primarily as a dissolved ion, Fe can be transported as ferrous (Fe^{2+}) and ferric (Fe^{3+}) ions and as suspended Fe(III) solids. In the presence of O_2, highly soluble Fe^{2+} tends to oxidize to relatively insoluble Fe^{3+} (reaction 2). At pH greater than 3, concentrations of Fe^{3+} tend to be limited by the formation of Fe(III) oxyhydroxides and related solids (reaction 3). The complete, stoichiometric oxidation of pyrite is indicated by combining reactions 1, 2, and 3. Half the protons (H^+), or acid, produced by the complete oxidation of pyrite result~~s~~ from the oxidation of pyritic sulfur to SO_4^{2-} (reaction 1) and the other half result~~s~~ from the oxidation of Fe^{2+} to Fe^{3+} and its hydrolysis and precipitation as $Fe(OH)_3$ (reactions 2 and 3).

The acid produced by pyrite oxidation or by hydrolysis of iron and other dissolved metals can be neutralized by reaction with calcite ($CaCO_3$), dolomite [$CaMg(CO_3)_2$], and other calcareous minerals as described below.

$$CaCO_3 + H^+ = Ca^{2+} + HCO_3^- \qquad (4)$$

$$CaMg(CO_3)_2 + 2\ H^+ = Ca^{2+} + Mg^{2+} + 2\ HCO_3^- \qquad (5)$$

These calcareous minerals are the dominant components of limestone and can occur in nodules, cementing agents, or fractures in sandstone, siltstone, shale, and associated strata of coal-bearing rocks. Alkalinity, represented by bicarbonate (HCO_3^-), and base cations, including calcium (Ca^{2+}) and magnesium (Mg^{2+}), are common products of neutralization by these calcareous minerals. Alkalinity also can be produced by bacterial reduction of sulfate in environments where organic matter is available and oxygen is deficient (Hedin and others, 1994; Langmuir, 1997). Where absent or deficient at a mine site, the addition of limestone or other alkalinity-producing materials to mine spoil or mine drainage can be effective for prevention or neutralization of AMD and the attenuation of dissolved metals. As the pH increases to near-neutral values (pH 6 to 7), concentrations of Fe^{3+}, Al^{3+}, and various other metals in AMD generally will decline; however, concentrations of SO_4^{2-}, Fe^{2+}, and Mn^{2+} generally will not be affected unless conditions become extremely reducing (Blowes and Ptacek, 1994; Langmuir, 1997; Cravotta and Trahan, 1999; Cravotta and others, 1999).

The pH of AMD can be unstable because of a general tendency for the exsolution of dissolved carbon dioxide (CO_2) and hydrogen sulfide (H_2S), the dissolution of O_2, and the consequent oxidation and hydrolysis of dissolved iron and manganese (Rose and Cravotta, 1998; Cravotta and others, 1999; Cravotta and Kirby, 2004b). AMD that initially has near-neutral pH and contains alkalinity ultimately could have acidic pH (less than 4.5) after its complete oxidation. This acidity results because of the oxidation and (or) hydrolysis of dissolved iron (reactions 2 and 3), manganese, aluminum, and other cations (Cravotta and Kirby, 2004b). Net-alkaline samples (alkalinity greater than acidity) tend to remain near neutral; however, net-acidic samples (acidity greater than alkalinity) tend to have acidic pH after complete oxidation. Hence, the characterization of AMD as acidic or neutral and the evaluation of appropriate remediation should consider the pH, acidity, alkalinity, and concentrations of dissolved metals.

Although sewage-treatment facilities serve the larger municipalities in the study area, such as Frackville, Ashland, Girardville, and Shenandoah, direct discharges of raw sewage and leaky on-lot sewage disposal systems degrade local stream sections throughout the basin (Pennsylvania Department of Environmental Protection, 2001, 2002). Agricultural practices affect the quality of streamwater in parts of the western subbasin, particularly Schwaben Creek (Pennsylvania Department of Environmental Protection, 2004). However, AMD from abandoned anthracite mines is the overwhelming source of contaminants to Mahanoy Creek, Little Mahanoy Creek, North Mahanoy Creek, Shenandoah Creek, Zerbe Run, and various smaller unnamed tributaries (Pennsylvania Department of Environmental Protection, 2001, 2002, 2004).

Water-Quality Protection and Restoration

Pennsylvania has adopted water-quality criteria intended to protect the anticipated uses of streams for (1) the maintenance and propagation of cold-water and warm-water fish; (2) water supply for domestic, industrial, livestock, wildlife, and irrigation purposes; (3) boating, fishing, and water-contact sports; (4) power; and (5) treated waste assimilation (Commonwealth of Pennsylvania, 2002). The main stem Mahanoy Creek is designated a warm-water fishery (WWF) and its tributaries are designated cold-water fisheries (CWF) (Commonwealth of Pennsylvania, 2002). To meet the WWF and CWF designations, the following criteria must be met (Commonwealth of Pennsylvania, 2001a, 2002):

- temperature during July and August not to exceed 66°F (18.9°C) or 87°F (30.6°C) for CWF and WWF, respectively;
- dissolved oxygen concentration greater than 5.0 mg/L for CWF and 4.0 mg/L for WWF;
- alkalinity not less than 20 mg/L as $CaCO_3$, except where natural conditions are less;
- pH not less than 6.0 or greater than 9.0;
- total iron concentration not to exceed 1.5 mg/L as a 30-day average;
- dissolved iron concentration not to exceed 0.3 mg/L;
- total manganese concentration not to exceed 1.0 mg/L; and
- total aluminum concentration not to exceed 0.75 mg/L.

The above criteria for chemical constituents have been incorporated in recently developed “total maximum daily loads” (TMDLs) for Mahanoy Creek and its tributaries (Pennsylvania Department of Environmental Protection, 2002; Dempsey and others, 2002). The intent of the TMDLs is to identify the amount of a contaminant that a stream can assimilate without exceeding its water-quality standards. TMDLs have been calculated for each of the above contaminants documented as causing impairment. However, the criteria for TMDLs are limited to only some of the constituents that may have adverse effects to aquatic organisms or humans. Trace metals such as arsenic (As), cadmium (Cd), cobalt (Co), copper (Cu), lead (Pb), nickel (Ni), and zinc (Zn) commonly are found in AMD at levels that are above background concentrations and may be toxic (Elder, 1988; Hyman and Watzlaf, 1997; Rose and Cravotta, 1998; Cravotta and others, 2001). Guidelines for the protection of freshwater aquatic organisms and human health from these trace metals and other contaminants have been established by the U.S. Environmental Protection Agency (1997, 2002a, 2002b) and adopted by the Commonwealth of Pennsylvania (2001b).

The restoration of water quality in mining-affected areas generally involves a combination of remining, land reclamation, and AMD treatment. Remining includes reprocessing of culm and silt to separate previously uneconomic coal from waste rock and surface mining or “daylighting” of old underground mine complexes. Such remining currently is done under stricter environmental regulations than in the past and can result in reduced infiltration to underground mines and (or) improved water quality. Land reclamation can involve the filling of open pits or shafts, the removal and (or) revegetation of coal spoil or culm banks, and the restoration of stream channels and streamflow.

Treatment of AMD may be necessary to neutralize acidity and remove dissolved and suspended metals from the hydrologic system where reclamation of a mine or mining-related surface effects is not possible. The conventional treatment for metal-laden effluent that has excess acidity involves aeration and the addition of strong alkaline chemicals (Skousen and others, 1998). Although effective, this “active” treatment approach can be expensive because of the high cost of chemical reagents, operation, and maintenance. Alternative treatment methods for AMD include wetlands and limestone-based systems (Hedin and others, 1994; Skousen and others, 1998; Watzlaf and others, 2000). Generally, for net-acidic AMD, limestone-based treatments such as an open limestone channel (OLC), anoxic limestone drain (ALD), oxic limestone drain (OLD), or vertical flow compost wetland (VFCW) also known as “successive alkalinity producing systems” (SAPS) could be appropriate to add alkalinity (fig. 2). For net-alkaline AMD or after treatment with alkalinity producing systems, oxidation ponds or aerobic wetlands are useful to remove metals as solids. These “passive” treatment systems generally require little maintenance over their design life (typically 20 years) but are limited by slower rates of neutralization and contaminant removal and, consequently, may require larger land area than for conventional “active” treatments. Nevertheless, passive systems can be cost effective where water chemistry meets suggested criteria and land and component materials are available locally (Skousen and others, 1998).

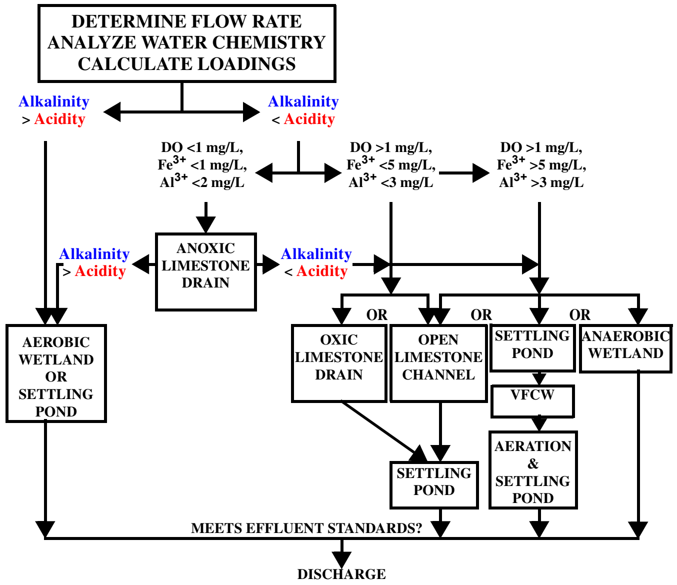

Figure 2. Flow chart for selection of passive-treatment alternatives modified from Hedin and others (1994), Skousen and others (1998), and Pennsylvania Department of Environmental Protection (1999). Vertical flow compost wetland (VFCW) also known as SAPS or RAPS. Less than (<), greater than (>), milligrams per liter (mg/L).

Operation Scarlift project SL-197 (Sanders & Thomas, Inc., 1975) identified 31 mine discharges and presented 3 abatement plans that included mine sealing, backfilling of strip pits, regrading and revegetating steep banks, relocation and lining of streams, and active chemical treatment to remediate AMD and erosion problems in the upper Mahanoy Creek Basin. The estimated cost in 1975 dollars for these plans ranged from $6.4 to $31.3 million (Sanders & Thomas, Inc., 1975). However, because these proposals were too costly or impractical, little has been done to mitigate the AMD.

Major changes could have occurred in the flow and quality of the mine discharges and in the quality of Mahanoy Creek in the decades since the Operation Scarlift report was completed in 1975 (Reed and others, 1987; Wood, 1996). Furthermore, passive-treatment systems that recently have been developed could be lower cost alternatives to remediate AMD compared to active treatment methods considered previously. This study was undertaken to provide an updated assessment of the hydrological and chemical characteristics of the AMD and hydrological, chemical, and biological characteristics of the receiving streams in the basin and to determine the ecological effects, priorities, and alternatives for AMD remediation.

Methods of Water-Quality Site Selection, Sampling, and Analysis

Before the basin-wide synoptic monitoring was initiated, published data on the locations of AMD and stream monitoring sites as reported by Sanders & Thomas, Inc. (1975), Reed and others (1987), and Wood (1996) were reviewed, compiled, and mapped. The mapped locations were then compared with electronic images of mine maps and original hand-marked copies of topographic maps provided by L.A. Reed of USGS to resolve discrepancies in site names or locations.

A total of 35 AMD sites were identified from various documents; 30 of these sites were sampled (tables 1 and 2). Each of the AMD sites is identified by local identification numbers from 1 to 35, in approximate downstream order, with the prefix M in column 1 of table 1 (fig. 1). Four of the Scarlift sample sites (M06, M10, M16, and M35) and two of the previous USGS sample sites (M09 and M15) could not be located or were not accessible for this study. The Packer #5 Borehole and Breach (M14) previously sampled by Reed and others (1987) is along an open ditch more than 0.25 mi (0.40 km) downstream from the Packer #5 Borehole (M12) and Breach (M13). Because mine water sources are sampled at this site, it was assigned the local identification number M14. However, because of its physical setting, site M14 was listed with stream sites and was assigned a downstream-order station number in table 2. Additionally, 31 stream sites were surveyed, identified by local identification numbers from 1 to 31 with the prefix S in column 1 of table 2 (fig. 1). A formal USGS station identification number was assigned to each of the sampled AMD and stream sites for their incorporation in the USGS National Water Information System (NWIS) database (tables 1 and 2). Generally, previously published latitude and longitude of Reed and others (1987) were used to assign the corresponding station numbers for AMD sites (table 1). In some cases, these station numbers differ from the reported coordinates because of typographical errors in the previously published coordinates and (or) inconsistency between the current global positioning system (GPS) measurement and previously mapped locations. Stream or other surface-water sites were assigned USGS station numbers based on the relative downstream order within a given watershed (table 2).

Water-quality and flow data for the synoptic surveys were collected at the AMD and stream sites during high base-flow conditions, March 26-28, 2001, and low base-flow conditions, August 20-22, 2001 (tables 1 and 2). Two teams of two persons each were deployed to assigned sites with GPS units and identical sets of water-quality sampling and monitoring equipment. When samples were collected, the flow rate was measured using a wading rod and pygmy meter or volumetrically (Rantz and others, 1982a, 1982b), and the temperature, specific conductance (SC), pH, redox potential (Eh), and dissolved oxygen (DO) concentration were measured using a field-calibrated YSI multiparameter water-quality sonde using methods described by Wood (1976) and U.S. Geological Survey (1997 to present). Water samples were collected as 1-L grab samples as close as possible to the discharge location or where streamflow was well mixed, avoiding bottom sediments and other debris. The grab samples were split into subsamples for field filtration and preservation as appropriate. Filtered (0.45-µm pore-size filter), unpreserved samples for analysis of major anions and unfiltered samples without head space for analysis of alkalinity and acidity were stored in sample-rinsed polyethylene bottles at 4°C. Samples for dissolved (0.45-µm pore-size filter) and total recoverable (whole-water; in-bottle nitric and hydrochloric acid digestion) metal analysis were acidified with nitric acid (HNO_3) and stored in acid-rinsed polyethylene bottles.

Within 48 hours of sampling, alkalinity was titrated in the laboratory with sulfuric acid to a fixed endpoint pH of 4.5. Concentrations of major anions, major cations, and trace elements were determined using ion chromatography (IC) and inductively coupled plasma emission mass spectrometry (ICP-MS) and inductively coupled plasma-atomic emission spectrometry (ICP-AES) (Fishman and Friedman, 1989; Fishman, 1993; Crock and others, 1999). The cation and trace-element analyses were completed by the USGS Mineral Resources Research Laboratory in Denver, Colo., and the anion analyses including sulfate, chloride, nitrate, and phosphate were completed at the Actlabs Laboratory in Toronto, Ontario. Data for most constituents were available for filtered and unfiltered samples and by different methods (ICP-MS, ICP-AES, and (or) IC). For example, sulfur (presumably sulfate) and phosphorus (presumably phosphate) were reported analytes by IC and ICP-MS. Generally, filtered samples had constituent concentrations less than or equal to the unfiltered samples (as discussed in more detail later in this report), and the results by different methods were comparable. However, the values for sulfate by IC typically were less than those by ICP-MS. Charge imbalances were calculated as the difference between cation and anion equivalents relative to the mean of cation and anion equivalents and routinely were less than 10 percent considering data for sulfate by IC. Hence, sulfate by IC is used hereinafter.

Bottom sediments were collected during low base-flow water-quality sampling in August 2001. At each stream and AMD site, a composite sample was collected using a polyethylene scoop for subsampling to a depth of approximately 1 in. (2.5 cm) at four to six points across the stream or discharge location, emphasizing depositional zones behind obstructions. Where present, ochreous coatings were scraped from rocks and other substrate materials. Sediments were sealed with associated "native" water in plastic bags and stored at 4°C until laboratory processing could be completed. In the laboratory, the composite sample for each site was wet-sieved with deionized water through a 100-mesh (0.182-mm) stainless-steel screen. The material smaller than 100 mesh was centrifuged for 1 hour at 10,000 revolutions per minute and then oven dried at 32°C. Dried sediments were disaggregated by crushing gently with a ceramic mortar and pestle and then analyzed for chemical and mineralogical composition. The dried powder was sieved to less than 37 µm and mounted in the 2.5-cm diameter well of a stainless-steel holder for mineralogical analysis by X-ray diffraction (XRD). The XRD patterns were collected on a Scintag theta-theta diffractometer using Cu radiation over the range 5° to 80° at a 1° per minute continuous scan rate. Major, minor, and trace elements in a 1-gram subsample of the dried sediment were determined by ICP-MS and ICP-AES after decomposition with a mixture of hydrochloric, nitric, perchloric, and hydrofluoric acids at low temperature (Fishman and Friedman, 1989; Fishman, 1993; Crock and others, 1999; Taggart, 2002).

Fish were collected by electrofishing over a 500-ft (150-m) reach consisting of mixed riffle, run, and pool habitats at selected stream sites, held for measurement and identification, checked for anomalies, and then released in accordance with methods described by Meador and others (1993a, 1993b), U.S. Environmental Protection Agency (1993), and Bilger and others (1999). Four of the sites (S10, S16, S23, and S30) on Mahanoy Creek were downstream of AMD sources; one site (S28) on Schwaben Creek was within an unmined area. Selected large specimens (greater than 25-cm length) of white sucker (*Catostomus commersoni*) from Mahanoy Creek near Gowen City (S23) were sacrificed for analysis of trace metals in whole fish. Six specimens were frozen for transport to the laboratory in accordance with preparation protocols (Hoffman, 1996). The whole-fish samples were homogenized to form a single composite, and a subsample was dried and acid-extracted for analysis of trace metals by ICP-AES, ICP-MS, or cold vapor-atomic absorption spectrophotometry (Hoffman, 1996).

Benthic macroinvertebrate surveys were conducted at most water-quality monitoring stations and additional sites during base-flow conditions in 2000-2002 by Martin A. Friday of PaDEP using rapid bioassessment protocols (U.S. Environmental Protection Agency, 1993; Barbour and others, 1999). Some of the benthic surveys were conducted in coordination with USGS water-quality sampling surveys in August 2001 and fish surveys in October 2001. A rectangular frame kicknet with 0.6-mm screen size was used to capture debris and organisms dislodged from the streambed. An area of approximately 0.5-m^2 was "kicked" upstream of the net for a total of 30 seconds for

Table 1. Description of abandoned mine drainage sites in the Mahanoy Creek Basin, Pennsylvania, surveyed in 2001.
[n.a., not applicable; latitude and longitude listed without degree, minute, and second symbols; 404916.6 represents 40°49'16.6" north latitude and 760617.4 represents 76°06'17.4" west longitude]

Site name and local identification number[1]	Published site identification number			Latitude	Longitude	Watershed[2]	Sample[3]	
	Scarlift site number[4]	WRI 85-4038[5]	U.S. Geological Survey station identification number[6]				High	Low
Vulcan-Buck Mountain Mine Morris Tunnel (M01)	1	61	404916076071701	404916.6	760617.4	UMC	Dry	Dry
Vulcan-Buck Mountain Mine seepage (M02)	n.a.	62	404858076072501	404857.7	760725.2	UMC	Yes	Dry
Vulcan-Buck Mountain Mine Boreholes (M03)	4	63	404855076073501	404855.6	760735.0	UMC	Yes	Yes
Gilberton Mine Pump (M04)	7	64	404801076123401	404759.1	761230.7	UMC	Yes	Dry
Weston Mine surface areas seepage (M05)	n.a.	66	404830076144901	404829.8	761447.9	SC	Yes	Dry
Weston Mine Lost Creek-Buck Mountain discharge (M06)	21	n.a.	n.a.	404851.0	761430.0	SC	No	No
Weston Mine Lost Creek Borehole (M07)	23	67	404825076144901	404828.3	761447.9	SC	Yes	Yes
Hammond Mine Seepage (M08)	25	68	404805076162001	404802.5	761620.9	SC	Yes	Dry
Hammond Mine Connerton Village Boreholes (M09)	26	69	404806076160401	404806.0	761604.0	SC	No	No
Hammond Mine Connerton #1 and #2 discharges (M10)	28-29	n.a.	n.a.	404748.0	761627.0	SC	No	No
Girard Mine seepage (M11)	10-13	70	404730076160601	404731.2	761605.9	MMC	Yes	Yes
Packer #5 Mine Borehole (M12)	15	71A	404740076162201	404741.1	761621.8	MMC	Yes	Yes
Packer #5 Mine Breach (M13)	16	71B	404739076162801	404739.9	761629.5	MMC	Yes	Yes
Preston Mine Preston #3 water level drift (M15)	n.a.	72	404744076163001	404744.0	761630.0	MMC	No	No
Preston Mine #2 overflow (M16)	31	n.a.	n.a.	404745.0	761708.0	MMC	No	No
Preston Mine #3 Tunnel overflow (M17)	32	73	404725076173401	404724.6	761732.8	MMC	Yes	Yes
Bast Mine Tunnel (M18)	34	74	404729076180801	404730.9	761804.6	MMC	Yes	Yes
Centralia Mine Tunnel (M19)	36	75	404727076192601	404735.3	761925.7	MMC	Yes	Yes
Bast Mine overflow (M20)	n.a.	76	404711076190901	404711.1	761907.5	MMC	Yes	Dry
Bast Mine Oakland Tunnel (M21)	38	77	404706076195401	404706.0	761952.5	MMC	Yes	Yes
Tunnel Mine seepage to ditch from spoil bank/pond (M22)	n.a.	n.a.	404655076195301	404655.0	761953.2	MMC	No	Yes
Tunnel Mine discharge from spoil bank/pond (M23)	n.a.	n.a.	404650076200201	404649.6	762001.9	MMC	No	Yes
Tunnel Mine drain pool area and storage (M24)	40-41	78	404645076201201	404644.8	762011.5	MMC	Yes	Yes
Tunnel Mine Orchard Drift overflow near Ashland sewage (M25)	42	n.a.	404648076202301	404648.0	762023.4	MMC	No	Yes
Potts Mine West Breach (M26)	48	79	404634076221901	404633.3	762216.8	MMC	Yes	Yes
Potts Mine East Breach (M27)	49	80	404624076221501	404628.7	762215.8	MMC	Yes	Yes
Lavelle Mine Lavelle slope (M28)	53	91	404558076240501	404558.0	762405.0	MC	Yes	Yes
Locust Gap Mine Locust Gap Tunnel (M29)	54	92	404504076261201	404504.0	762612.0	MC	Yes	Yes
Locust Gap Mine Helfenstein Tunnel (M30)	n.a.	n.a.	404515076265201	404515.1	762652.1	MC	Yes	Yes
Locust Gap Mine Doutyville Tunnel (M31)	56	94	404435076283801	404435.0	762838.0	MC	Yes	Yes
N. Franklin Mine drift and Borehole (M32)	62	111	404617076404401	404616.5	764042.1	ZR	Yes	Yes
N. Franklin Mine seepage (M33)	63	112	404636076405801	404640.7	764058.6	ZR	Yes	Yes
N. Franklin Mine bank seepage (M34)	n.a.	n.a.	404617076405201	404617.4	764051.7	ZR	Yes	No
N. Franklin Mine Sunshine Mine overflow (M35)	65	n.a.	n.a.	404552.0	764351.0	ZR	No	No

[1] Local site identification numbers, in parentheses, used to indicate site location in figure 1.

[2] Streams are shown in figure 1. UMC=Upper Mahanoy Creek; MMC=Middle Mahanoy Creek; MC=Mahanoy Creek; NMC=North Mahanoy Creek; SC=Shenandoah Creek; ZR=Zerbe Run.

[3] Water-quality and flow data collected for high base-flow conditions on March 26-28, 2001, and (or) low base-flow conditions on August 20-22, 2001. Streambed chemistry samples also collected at surface-water and ground-water stations during the low base-flow survey.

[4] Scarlift site number of Sanders & Thomas, Inc. (1975).

[5] Informal USGS site number of Reed and others (1987).

[6] Formal USGS station identification number was assigned only if the site was sampled by USGS for this study or Reed and others (1987). Latitude and longitude for sites that were not sampled by USGS were estimated from maps in Sanders & Thomas, Inc. (1975).

Table 2. Description of stream sites in the Mahanoy Creek Basin, Pennsylvania, surveyed in 2001.
[Cr, Creek; Unn., Unnamed; Trib., Tributary; nr near; bl, below; n.a., not applicable; latitude and longitude listed without degree, minute, and second symbols; 404934.9 represents 40°49'34.9" north latitude and 760521.4 represents 76°05'21.4" west longitude]

Site name and local identification number[1,2]	Abandonded mine drainage sources[3]	Scarlift site number[4]	U.S. Geological Survey station identification number	Latitude	Longitude	Sample[5]			
						High	Low	Fish	Bugs
Mahanoy Cr nr Buck Mountain (S01)	n.a.	n.a.	0155521012	404934.9	760521.4	No	Dry	No	No
Unn. Trib. to Mahanoy Cr nr Mahanoy City (S02)	n.a.	n.a.	n.a.	404856.7	760723.5	Yes	Dry	No	No
Mahanoy Cr nr Mahanoy City (S03)	M01	n.a.	0155521021	404856.7	760723.5	Yes	Dry	No	Yes
Mahanoy Cr at Mahanoy City (S04)	(S01),M02	n.a.	0155521023	404855.8	760731.2	Yes	Dry	No	No
Unn. Trib. to N. Mahanoy Cr nr Shoemakers (S05)	n.a.	n.a.	0155521037	405008.0	760637.0	No	Yes	No	No
N. Mahanoy Cr at Mahanoy City (S06)	n.a.	3	0155521049	404856.2	760825.8	Yes	Dry	No	Yes
Waste House Run at Yatesville (S07)	n.a.	6	0155521080	404956.0	760950.0	Yes	Yes	No	Yes
Waste House Run at St. Nicholas (S08)	n.a.	n.a.	01555211	404817.2	761036.8	Dry	Dry	No	No
Mahanoy Cr at Gilberton (S09)	(S01-S08),M03	8	0155521140	404759.0	761230.0	No	Yes	No	Yes
Mahanoy Cr at Girardville (S10)	(S01-S09),M04,M11	14	01555212	404732.0	761624.0	Yes	Yes	Yes	Yes
Kehly Run nr Shenandoah (S11)	n.a.	n.a.	0155521206	405009.9	761149.2	Yes	Yes	No	Yes
Lost Cr nr Lost Creek (S12)	M06	22	0155521332	404853.3	761430.4	Yes	Yes	No	Yes
Shenandoah Cr at Lost Creek (S13)	(S11,S12)	24	0155521334	404824.6	761448.4	Yes	Yes	No	Yes
Shenandoah Cr nr Girardville (S14)	(S13)M05-M10	30	0155521339	404745.1	761632.6	Yes	Yes	No	Yes
Unn. Trib. to Mahanoy Cr "Big Mine Run" nr Girardville (S15)	M19	n.a.	0155521346	404714.4	761904.5	Yes	Yes	No	No
Packer #5 Mine Breach and Borehole at Mahanoy Cr (M14)[b]	M12-M13	n.a.	0155521341	404739.9	761643.3	Yes	Yes	No	No
Mahanoy Cr nr Ashland (S16)	(S10,S14)M14-M25	43	0155521356	404635.1	762021.7	Yes	Yes	Yes	Yes
Mahanoy Cr nr Gordon (S17)	(S16)	n.a.	0155521369	404524.3	762024.9	Yes	Yes	No	No
Rattling Run at Gordon (S18)	n.a.	45	0155521486	404457.5	762016.7	Yes	Yes	No	No
Little Mahanoy Cr at Gordon (S19)	(S18)	46	0155521488	404514.6	762032.8	Yes	Yes	No	Yes
Unn. Trib. to Mahanoy Cr "Big Run" nr Lavelle (S20)	M26,M27	51	01555228	404512.9	762221.2	Yes	Yes	No	Yes
Crab Run nr Taylorville (S21)	n.a.	n.a.	01555232	404446.4	762329.4	No	Yes	No	Yes
Mahanoy Cr nr Lavelle (S22)	(S17,S19,S20,S21)	52	01555234	404509.0	762350.0	Yes	Yes	No	No
Mahanoy Cr nr Gowen City (S23)	(S22)M28-M31	58	01555240	404425.4	763255.1	Yes	Yes	Yes	Yes
Zerbe Run at Trevorton (S24)	n.a.	61	0155524529	404652.8	764058.1	Yes	Yes	No	Yes
Unn. Trib. to Zerbe Run at Trevorton (S25)	M32-M34	n.a.	0155524558	404646.4	764101.3	Yes	Yes	No	Yes
Zerbe Run nr Dornsife (S26)	(S24,S25)M35	66	01555246	404521.0	764517.0	Yes	Yes	No	Yes
Mahanoy Cr nr Dornsife (S27)	(S23,S26)	67	0155525004	404354.0	764738.7	Yes	Yes	No	No
Schwaben Cr nr Red Cross (S28)	n.a.	68	0155525061	404257.7	764645.9	Yes	Yes	Yes	Yes
Mouse Cr bl Urban (S29)	n.a.	70	0155525069	404122.0	764617.1	Yes	Yes	No	Yes
Mahanoy Cr at Kneass (S30)	(S27,S28,S29)	71	01555251	404328.8	764855.4	Yes	Yes	Yes	Yes
Mahanoy Cr nr Herndon (S31)	(S31)	n.a.	01555252	404334.0	765016.0	No	Yes	No	No

1. Local site identification numbers, in parentheses, used to indicate site location in figure 1.

2. The Packer #5 borehole and breach (M14) was assigned a surface-water station number (0155521341) and is treated as surface water in subsequent figures and tables. However, this site consists entirely of mine drainage from the Packer #5 borehole (M12) and breach (M13) and, hence, was given a local number with prefix "M".

3. AMD sources identified by local site numbers in table 1. If the site is downstream from another stream monitoring site(s), the AMD associated with the upstream site, in parentheses, also applies.

4. Scarlift site numbers based on report of Sanders & Thomas, Inc. (1975).

5. Water-quality and flow data collected for high base-flow conditions on March 26-28, 2001, and (or) low base-flow conditions on August 20-22, 2001. Streambed chemistry samples collected at all surface-water and ground-water stations during August 2001. Fish were collected during low base-flow conditions on October 10-11, 2001. Macroinvertebrates, or "bugs," collected and identified by Martin A. Friday of the Pennsylvania Department of Environmental Protection during July 2000 to May 2002 at sites indicated plus additional sites in the watershed.

each sample. Samples were collected from three habitats consisting of shallow riffle with exposed cobbles, deeper riffle, and run habitats. The samples were composited in the field and identified to family level. Counts of organisms for various taxa were recorded as rare (1-2), present (3-9), common (10-24), abundant (25-100), and very abundant (greater than 100). By assuming the average value for the abundance range indicated for each taxa and a value of 125 for very abundant taxa, the semi-quantitative taxa abundance data were used to calculate the family-level biotic index (Hilsenhoff, 1988; Barbour and others, 1999).

The data assembled for the assessment were incorporated into digital databases, including the USGS NWIS, spreadsheets, and a GIS. These data are summarized by site in appendix A. The water-quality data also are accessible on the World Wide Web for selected stations (tables 1 and 2) and dates on the USGS NWIS (*http://wwwpah2o.er.usgs.gov/*).

Priority ranks of AMD sites were determined and compared on the basis of ranks of contaminant loads during low and high base-flow conditions. Because acidity of AMD is largely a function of the pH and dissolved metal concentrations, prioritization methods evaluated metals loading for ranking the AMD sources using an approach similar to that of Cravotta and Kirby (2004a). The metals loading was computed as the product of flow rate and the sum of concentrations of the dissolved metals as

$$\text{Metal load} = f \cdot Q \cdot (C_{Fe} + C_{Mn} + C_{Al}), \qquad (6)$$

where Q is the instantaneous flow rate in cubic feet per second and C with a subscript Fe, Mn, or Al indicates dissolved iron, manganese, or aluminum concentration, respectively, in milligrams per liter. For these units of measure, the conversion factor, f = 0.893, yields annual loading in megagrams (Mg/yr), whereas f = 0.984 yields annual loading in short tons (ton/yr).

Acidity for all the samples was computed from pH and dissolved metals concentrations in milligrams per liter as

$$\text{Acidity}_{\text{computed}}\ (\text{mg/L CaCO}_3) = 50 \cdot (10^{(3\text{-pH})} + 2 \cdot C_{Fe}/55.85 + 2 \cdot C_{Mn}/54.94 + 3 \cdot C_{Al}/26.98). \qquad (7)$$

The computed acidity avoids issues of different analytical methods, the lack of reporting negative values for the “hot” acidity, or an assumed value of zero for near-neutral pH samples; however, the computed acidity also involves assumptions regarding valence or speciation of the dissolved metals (Kirby and Cravotta, 2004; Cravotta and Kirby, 2004b). The net alkalinity was computed by subtracting the computed acidity from measured alkalinity as

$$\text{Net alkalinity (mg/L CaCO}_3) = \text{Alkalinity} - \text{Acidity}_{\text{computed}}. \qquad (8)$$

The net alkalinity on the basis of computed acidity (equations 7 and 8) is comparable to the measured “hot” acidity where acid added at the start of the titration is subtracted from the total base consumed and negative values of the acidity for high-pH samples are reported (Cravotta and Kirby, 2004b).

The physical setting (area available for treatment, slope, access), maximum measured flow, and specific water-quality data for each AMD source including minimum net alkalinity (equation 8) and maximum concentrations of dissolved oxygen and metals for the low and high base-flow samples were used to determine feasibility of remediation. Guidelines similar to those introduced by Hedin and others (1994) and modified by Skousen and others (1998) were used for the identification of appropriate remedial alternatives (fig. 2). Consideration was given to remining or removal of culm banks, various passive-treatment technologies, active treatment, and the “no-action” alternative. Computed wetland size, based on the 180 lb/acre/d (20 g/m^2/d) iron-loading rate of Hedin and others (1994), was compared with available land area (not considering ownership) to indicate feasibility for implementation of passive treatment at each site.

Effects of Abandoned Mine Drainage in the Mahanoy Creek Basin

Data for flow rate, pH, acidity, net alkalinity, and metals concentrations in samples from each AMD and stream site surveyed for the Mahanoy Creek watershed assessment during high base-flow conditions in March 2001 and low base-flow conditions in August 2001 are summarized in table 3 and figure 3. These high and low base-flow data were collected with the intent to bracket average conditions and indicate some of the variability in flow and chemistry at each AMD and stream monitoring site. All data on the flow rates, pH, concentrations of major ions, trace elements, nutrients, and other constituents are documented in appendix A.

Instantaneous streamflow ranged from 0 to 320 ft^3/s (0 to 544,000 L/min) (table 3); the medians for the low and high base-flow surveys were 2.0 and 9.3 ft^3/s (3,380 to 15,600 L/min), respectively. Flow rates of AMD ranged from 0 to 17.2 ft^3/s (0 to 29,200 L/min); the medians for the low and high base-flow surveys were 0.09 and 0.53 ft^3/s (153 to 900 L/min), respectively. With few exceptions, the flow rate at each stream and AMD site during March 2001 exceeded that during August 2001 (fig. 3). Generally, concentrations of sulfate, acidity, and dissolved iron and manganese at a particular site were greater at low base-flow conditions than high base-flow conditions; however, the pH and dissolved aluminum concentrations at a given site were comparable for low base-flow and high base-flow conditions (table 3, fig. 3).

Most AMD and streamwater samples collected during the high base-flow and low base-flow surveys had near neutral pH (table 3); the median pH for AMD was 6.0 and for streamwater was 6.8. The AMD and streamwater samples that had pH less than 5 were net acidic and those that had pH greater than 6 were net alkaline (table 3). The low-pH samples had elevated concentrations of acidity, metals (Fe, Mn, Al), and sulfate. Many of the high-pH samples had elevated concentrations of alkalinity, sulfate, calcium, and magnesium (table 3, appendix A),

Table 3. Values of selected constituents for streamwater and abandoned mine drainage samples collected for the Mahanoy Creek, Pennslyvania, watershed assessment during March 26-28 and August 20-22, 2001.

[ID No., identification number; 0, not flowing or dry; n.d., no data; <, less than; ft^3/s, cubic feet per second; L/min, liters per minute; mg/L, milligrams per liter; Mg/yr, megagrams per year; µm, micrometer]

Local site ID No.[1]	Flow rate (ft^3/s)[2]		Flow rate (L/min)		Dissolved oxygen (mg/L)		pH		Net alkalinity[3] (mg/L $CaCO_3$)		Dissolved[4] iron (mg/L Fe)		Dissolved aluminum (mg/L Al)		Dissolved manganese (mg/L Mn)		Dissolved sulfate (mg/L SO_4)		Dissolved nitrate (mg/L N)		Dissolved phosphorus (mg/L P)		Dissolved Fe, Al, and Mn load (Mg/yr)	
	Mar	Aug	Mar	Aug	Mar	Aug	Mar	Aug	Mar	Aug	Mar	Aug	Mar	Aug	Mar	Aug	Mar	Aug	Mar	Aug	Mar	Aug	Mar	Aug
Streamwater (SW, SW-AMD)																								
S01	n.d.	0	n.d.	0	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0
S03	0.67	0	1,140	0	10.1	n.d.	4.0	n.d.	-11	n.d.	0.04	n.d.	0.96	n.d.	0.31	n.d.	34	n.d.	0.40	n.d.	<0.01	n.d.	0.8	0
S04	8.90	0	15,100	0	2.5	n.d.	4.8	n.d.	-20	n.d.	3.2	n.d.	2.0	n.d.	1.1	n.d.	105	n.d.	.40	n.d.	<.01	n.d.	50.1	0
S05	n.d.	0.04	n.d.	68	n.d.	7.9	n.d.	4.9	n.d.	-1	n.d.	<0.01	n.d.	0.09	n.d.	0.06	n.d.	6	n.d.	<.01	n.d.	<.01	n.d.	<.1
S06	8.70	0	14,800	0	11.1	n.d.	5.6	n.d.	1	n.d.	.01	n.d.	.06	n.d.	.02	n.d.	5	n.d.	.05	n.d.	.01	n.d.	.8	0
S07	1.33	.11	2,260	187	10.8	6.5	6.1	6.5	3	6	.01	.02	.03	.03	.01	.04	4	5	.04	.06	<.01	<.01	.1	<.1
S08	0	0	0	0	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0	0
S09	n.d.	2.80	n.d.	4,760	n.d.	6.7	n.d.	5.0	n.d.	-6	n.d.	.41	n.d.	.21	n.d.	1.8	n.d.	256	n.d.	2.0	n.d.	.06	n.d.	6.1
S10	32.1	7.84	54,500	13,300	9.1	6.9	5.9	6.3	-3	9	2.9	6.8	.18	.01	1.7	2.6	138	203	.11	.66	.10	<.01	137	65.9
S11	1.56	.17	2,650	289	11.3	6.8	4.7	4.8	-3	-2	.08	.18	.43	.09	.10	.09	7	5	<.10	.05	<.01	<.01	.8	.1
S12	1.03	.11	1,750	187	12.2	7.5	6.2	6.3	3	3	.05	.03	.06	.02	.09	.18	10	12	.20	.13	.03	.02	.2	<.1
S13	3.00	.20	5,100	340	13.1	4.9	7.4	7.0	13	26	.41	.09	.05	.01	.87	1.8	74	166	2.0	3.3	.30	.33	3.6	.3
S14	12.1	5.50	20,600	9,340	9.8	5.8	6.7	6.9	43	68	3.1	.76	.01	.01	4.0	6.9	421	553	.60	.27	.03	.03	76.9	37.7
M14	10.27	9.42	17,400	16,000	3.3	2.0	6.4	6.4	95	76	13	21	.02	.01	6.2	9.2	597	595	<.1	<.05	<.01	<.01	176	254
S15	3.86	2.43	6,560	4,130	9.6	9.7	3.7	3.6	-60	-79	5.8	3.1	5.6	8.9	4.8	6.0	482	524	.50	<.10	<.01	<.01	55.9	39.1
S16	87.7	50.2	149,000	85,300	9.4	9.0	7.0	7.0	35	42	4.8	5.7	.02	.02	3.4	6.5	321	491	.16	.30	.03	.04	644	548
S17	112	46.2	190,000	78,500	9.7	8.3	7.0	6.9	32	51	4.2	2.1	.02	.04	3.3	6.0	295	480	.30	.45	.04	.04	753	336
S18	9.61	1.99	16,300	3,380	12.3	8.3	5.7	6.5	3	n.d.	.01	.01	.14	.01	.09	.01	7	6	.10	.18	.03	<.01	2.1	<.1
S19	30.1	6.80	51,100	11,600	13.1	8.3	7.0	7.0	11	20	.10	.08	.11	.01	.05	.02	10	11	.55	.52	.03	.07	7	.7
S20	6.08	1.33	10,300	2,260	10.6	9.6	8.2	8.0	108	178	.12	.22	.03	.01	.90	.75	286	508	.80	.50	<.01	<.01	5.7	1.2
S21	n.d.	.61	n.d.	1,040	n.d.	7.2	n.d.	7.1	n.d.	32	n.d.	.15	n.d.	.01	n.d.	.05	n.d.	9	n.d.	3.3	n.d.	.08	n.d.	.1
S22	107	51.4	182,000	87,300	10.4	8.4	7.0	7.6	23	47	1.6	.12	<.01	.04	2.3	4.6	227	427	.70	.76	.03	.04	373	219
S23	239	65.5	406,000	111,000	11.0	8.9	7.3	7.6	21	58	.76	.11	.01	.02	1.8	3.3	190	409	1.0	.73	.03	<.01	549	201
S24	7.99	.62	13,600	1,050	11.6	9.5	7.0	7.2	11	49	.14	.21	.04	.02	.06	.02	10	8	.51	.72	.01	.01	1.7	.1
S25	6.68	2.79	11,300	4,740	10.2	10.5	6.3	6.3	-6	-6	8.1	11	.05	.04	2	2.4	253	304	.07	<.01	<.01	<.01	60.6	33.5
S26	30.2	4.36	51,300	7,410	11.5	8.8	6.0	4.3	-3	-20	1.5	.15	.10	2.3	.93	2.7	99	264	.50	.60	.04	.02	68.2	20.1
S27	277	71.2	471,000	121,000	11.7	8.4	6.7	6.9	17	38	.28	.08	.02	.04	1.3	.90	152	382	.66	.80	.02	<.01	396	65.1
S28	28.4	1.66	48,300	2,820	12.6	8.4	7.3	7.5	14	54	.03	.06	.01	.02	.01	.02	12	11	6.0	1.9	.01	.03	1.4	.1
S29	4.28	.52	7,270	883	11.9	9.4	8.4	7.7	52	120	.05	.06	.02	.01	.01	.01	16	17	2.8	2.8	.01	.02	.3	<.1
S30	320	76.6	544,000	130,000	12.2	8.8	7.1	7.2	24	21	.11	.08	.01	.03	1.2	.54	132	363	1.8	.77	.01	<.01	378	44.7
S31	n.d.	76.3	n.d.	130,000	n.d.	8.7	n.d.	7.3	n.d.	43	n.d.	.08	n.d.	.02	n.d.	.23	n.d.	361	n.d.	.71	n.d.	<.01	n.d.	22.2
Abandoned Mine Drainage (GW, GW-AMD)																								
M01	0	0	0	0	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0	0
M02	8.79	0	14,900	0	0.2	n.d.	4.7	n.d.	-21	n.d.	3.4	n.d.	2.1	n.d.	1.2	n.d.	104	n.d.	0.01	n.d.	<0.01	n.d.	52.6	0
M03	5.28	2.64	8,970	4,490	.1	.2	4.9	4.8	-18	-18	3.6	6.3	1.6	1.1	1.2	1.2	105	111	.01	<0.01	<.01	<.01	30.2	20.3
M04	0	0	0	0	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0	0
M05	.01	0	17	0	9.7	n.d.	6.4	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0
M07	.38	.03	646	51	.3	.4	6.1	6.1	57	11	14	33	<.01	<.01	6.4	8.9	582	561	.40	<.05	<.01	<.01	6.9	1.2
M08	.27	0	459	0	.5	n.d.	6.6	n.d.	164	n.d.	5.2	n.d.	<.01	n.d.	3.5	n.d.	450	n.d.	.40	n.d.	<.01	n.d.	2.1	0
M11	4.10	2.73	6,970	4,640	2.2	.3	6.1	6.0	51	15	18	24	<.01	<.01	3.8	4.4	218	228	<.10	<.01	<.01	<.01	79.9	69.3
M12	5.27	3.62	8,950	6,150	.2	.3	6.3	6.3	108	96	12	21	<.01	<.01	5.9	9.0	600	596	.40	<.05	<.01	<.01	84.3	97.1
M13	5.00	5.80	8,500	9,850	.1	.4	6.3	6.2	95	94	13	22	.01	.01	6.3	9.2	600	595	<.10	<.10	<.01	<.01	86.3	162
M17	2.23	.67	3,790	1,140	.4	.2	6.3	6.2	59	72	4.8	4.7	.04	<.01	.95	.99	62	74	.01	<.01	<.01	<.01	11.5	3.4
M18	.67	.40	1,140	680	8.3	8.8	5.8	5.7	-2	-24	7.7	19	.35	.47	2.5	4.2	434	578	.40	<.05	<.01	<.01	6.3	8.5
M19	3.86	2.43	6,560	4,130	7.4	8.4	3.9	3.6	-59	-80	7.6	4.6	5.5	8.7	4.9	5.9	483	520	.40	<.05	<.01	<.01	62.1	41.7
M20	2.23	0	3,790	0	3.1	n.d.	6.5	n.d.	80	n.d.	8.4	n.d.	.04	n.d.	2.7	n.d.	372	n.d.	<.10	n.d.	<.01	n.d.	22.2	0
M21	4.00	2.30	6,800	3,910	3.8	5.2	6.3	6.4	74	79	7.3	14	.05	.08	2.6	3.1	360	351	<.10	<.01	<.01	<.01	35.6	35.3
M22	n.d.	.03	n.a.	51	n.d.	5.5	n.d.	4.1	n.d.	-109	n.d.	2.3	n.d.	15	n.d.	9.7	n.d.	696	n.d.	<.05	n.d.	<.01	n.d.	.8
M23	n.d.	.03	n.a.	51	n.d.	.1	n.d.	5.9	n.d.	55	n.d.	.03	n.d.	<.01	n.d.	2.7	n.d.	203	n.d.	<.01	n.d.	<.01	n.d.	.1
M24	.09	.13	153	221	8.9	.9	7.4	6.1	218	54	.24	14	<.01	.05	.82	2.7	181	221	<.10	<.01	<.01	<.01	.1	2
M25	n.d.	.04	n.a.	68	n.d.	1.5	n.d.	5.6	n.d.	21	n.d.	.04	n.d.	.17	n.d.	.82	n.d.	106	n.d.	<.01	n.d.	<.01	n.d.	<.1
M26	1.44	.36	2,450	612	.2	1.7	6.6	6.4	245	216	16	30	<.01	.24	3.6	5.1	787	785	<.10	<.05	.01	<.02	25.2	11.4
M27	.22	.29	374	493	.6	2.5	6.7	6.5	190	202	8.9	13	<.01	.01	2.1	2.3	329	347	<.10	<.05	<.01	<.01	2.2	4.0
M28	.23	.01	391	17	.5	1.0	3.9	4.3	-20	-32	2.2	10	1.3	1.5	1.6	1.8	77	94	.05	<.01	<.01	<.01	1.0	.1
M29	17.2	7.29	29,200	12,400	8.8	9.1	6.7	6.5	34	29	6.7	11	.05	.05	2.3	3.8	424	506	<.10	<.10	<.01	<.01	139	96.7
M30	.09	<.01	153	2	3.7	2.8	5.5	6.1	8	13	.01	1.9	<.01	.01	<.01	.93	18	20	.06	<.01	<.01	.01	<.1	<.1
M31	3.52	.99	5,980	1,680	9.3	9.6	5.0	6.1	-17	9	2.1	3.4	1.8	.34	1.3	1.5	181	239	<.10	<.01	<.01	<.01	16.4	4.6
M32	6.45	2.56	11,000	4,350	4.6	4.4	5.9	5.2	5	3	11	13	.32	.51	1.9	2.3	297	307	<.10	<.01	<.01	<.01	76.2	36.2
M33	.03	0	51	0	4.8	n.d.	3.1	n.d.	-110	n.d.	5.1	n.d.	9.8	n.d.	3.8	n.d.	419	n.d.	.10	n.d.	<.01	n.d.	.5	0
M34	.02	<.01	34	<2	.9	.6	2.9	2.9	-184	-220	21	31	14	17	3.0	3.9	454	506	<.10	<.10	<.01	<.01	.7	.1

1. Site descriptions given in tables 1 and 2. Site locations given in figure 1. Complete data for sampled sites given in appendix A.
2. For convenience, flow rate is given in English and standard international units; 1 ft^3/s = 1,699 L/min.
3. Net alkalinity = measured alkalinity - computed acidity per equations 7 and 8.
4. Concentration and loading data are for “dissolved” metals and other constituents in filtered (0.45-µm pore size) samples.

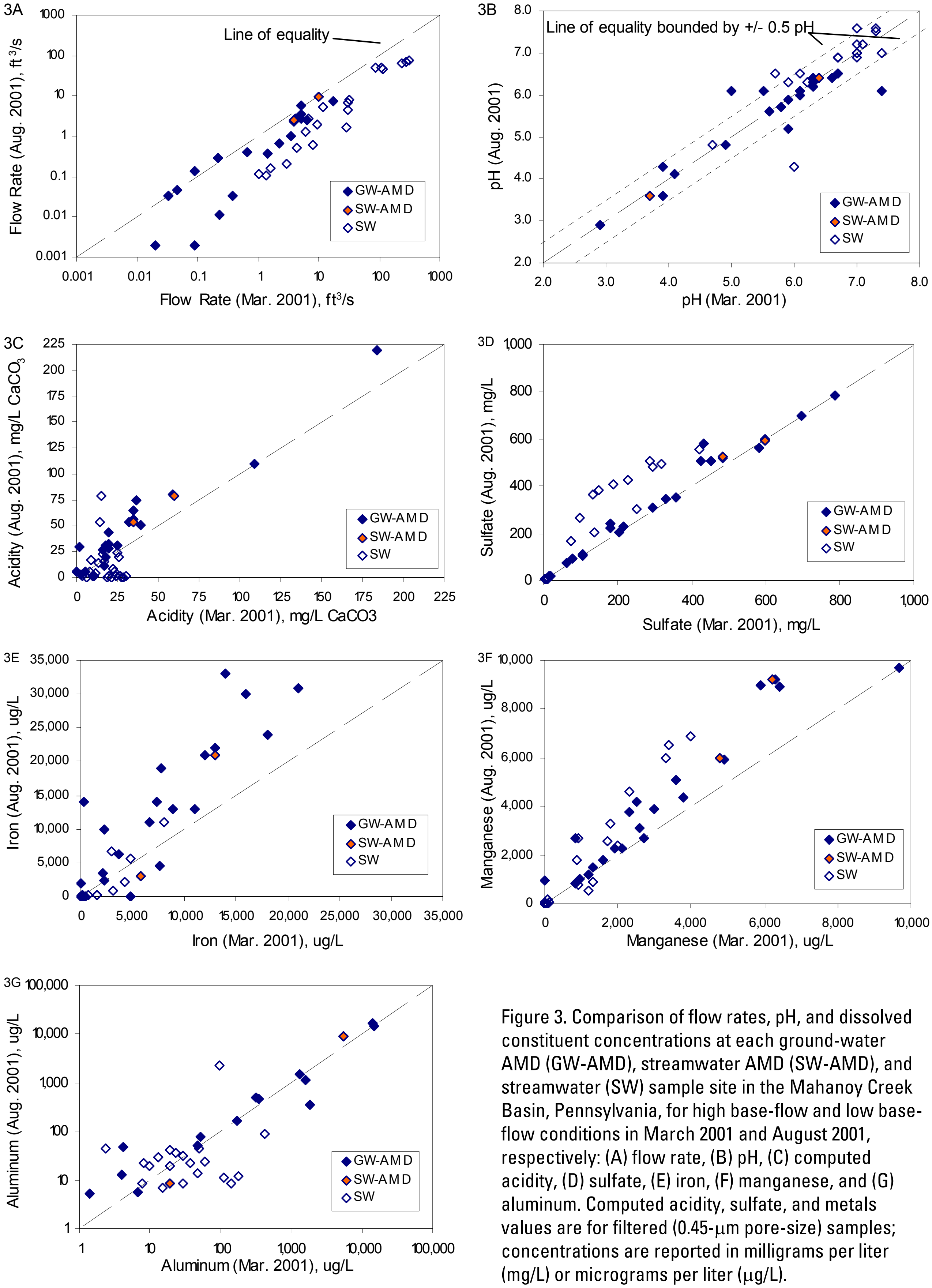

Figure 3. Comparison of flow rates, pH, and dissolved constituent concentrations at each ground-water AMD (GW-AMD), streamwater AMD (SW-AMD), and streamwater (SW) sample site in the Mahanoy Creek Basin, Pennsylvania, for high base-flow and low base-flow conditions in March 2001 and August 2001, respectively: (A) flow rate, (B) pH, (C) computed acidity, (D) sulfate, (E) iron, (F) manganese, and (G) aluminum. Computed acidity, sulfate, and metals values are for filtered (0.45-µm pore-size) samples; concentrations are reported in milligrams per liter (mg/L) or micrograms per liter (µg/L).

indicating their origin as acidic AMD that had been neutralized. Few AMD or streamwater samples for this survey had pH less than 4. The minimum pH value of 2.9 for AMD was observed at the North Franklin Mine (M34) during low and high base-flow conditions. The maximum pH value for AMD samples was 7.4 for high base-flow and 6.5 for low base-flow conditions. The minimum and maximum pH values for streamwater samples were 3.6 and 8.4, respectively. Notably, one stream site, Zerbe Run near Dornsife (S26), had acidic pH during low base-flow conditions but near-neutral pH during high base-flow conditions. The low pH and increased acidity (decreased net alkalinity) at this site for low base-flow conditions (table 3, fig. 3) can be attributed to various possible factors, including less dilution with alkaline water sources, increased rate of pyrite oxidation associated with increased air flow to the subsurface as the water table declined during summer drought, increased rate of Fe^{2+} oxidation associated with warmer temperature, and more complete oxidation and hydrolysis associated with longer transport or detention times at slower flow rates.

Although criteria for water-quality protection apply to the total concentration of regulated constituents (Commonwealth of Pennsylvania, 1998a, 1998b, 2001a, 2002), the dissolved concentrations generally are considered to be bioavailable (Elder, 1988; U.S. Environmental Protection Agency, 2002a). Thus, the "dissolved concentration" data for the filtered (0.45-µm pore size) samples can be evaluated directly with respect to aquatic-toxicity criteria. Furthermore, the dissolved metals loading is required for the selection and sizing of AMD treatment systems (fig. 2) and, as indicated by equations 7 and 8, the dissolved metals concentrations are included in the measurement and evaluation of acidity and net alkalinity. Thus, hereinafter, dissolved metals concentrations and corresponding computed values for metals loading, acidity, and net alkalinity will be emphasized. A subset of the metals analyzed for this study that have established criteria for protection of aquatic life or human health are described below.

Computed acidity (equation 7) of AMD samples ranged from 0 to 220 mg/L; corresponding net alkalinities (equation 8) of AMD ranged from 245 to -220 mg/L. Generally, because of dilution, the maximum values of acidity and net alkalinity were smaller for streamwater compared to the AMD sources. Although acidity at many AMD and streamwater sites was greater during low base-flow than high base-flow conditions (table 3, fig. 3), most AMD and streamwater sources consistently were net acidic (alkalinity less than acidity) or net alkaline (alkalinity greater than acidity) (table 3). The net acidic samples had elevated concentrations of dissolved iron as Fe^{2+}, manganese as Mn^{2+}, and sulfate.

Sulfate concentrations in the low and high base-flow AMD samples ranged from 18 to 787 mg/L (table 3, fig. 3). Sulfate concentrations in streamwater samples did not exceed 600 mg/L. The maximum concentrations of sulfate in AMD or streamwater samples were comparable for low base-flow and high base-flow conditions. Generally, streamwater samples had lower concentrations of sulfate than AMD sources with equivalent flow. Although the sulfate concentrations for a particular AMD source generally were comparable during high base-flow and low base-flow conditions, sulfate concentrations for streamwater sites generally were smaller during high base-flow than low base-flow conditions (fig. 3). Approximately half the AMD and streamwater samples exceeded the secondary drinking-water standard for sulfate (250 mg/L; U.S. Environmental Protection Agency, 2002b).

Iron concentrations in the low and high base-flow AMD samples ranged from 0.01 to 33 mg/L, and in streamwater samples ranged from less than 0.01 to 21 mg/L (fig. 3, table 3). With few exceptions, iron concentrations for a particular site were similar or greater during low base-flow than high base-flow conditions. Generally, surface water had lower concentrations of iron and other metals than AMD with equivalent flow. For samples containing more than 6.0 mg/L (6,000 µg/L) iron, the concentrations of dissolved and total iron were equivalent (fig. 4); that is, dissolved iron was predominant over suspended iron-containing particles. Approximately half the AMD samples exceeded the Commonwealth of Pennsylvania (1998a, 1998b) criteria for instantaneous maximum concentration of iron in effluent from an active mine (7.0 mg/L) (fig. 4). More than one-third of streamwater samples exceeded the Commonwealth of Pennsylvania (2002) instream criteria for total recoverable iron (1.5 mg/L) and dissolved iron (0.3 mg/L) (fig. 4). Most AMD and half the streamwater samples exceeded continuous exposure criteria for protection of aquatic life from dissolved iron (1.0 mg/L) (U.S. Environmental Protection Agency, 2002a). All but one AMD and most streamwater samples exceeded the U.S. Environmental Protection Agency (2002b) secondary drinking-water standard for iron (0.3 mg/L).

Manganese concentrations in the low and high base-flow AMD samples ranged from less than 0.01 to 9.7 mg/L, and in streamwater samples ranged from 0.01 to 9.2 mg/L (fig. 3, table 3). With few exceptions, dissolved manganese concentrations at a particular site were greater during low base-flow than high base-flow conditions (fig. 3). Generally, manganese was predominant as a dissolved constituent in the AMD and streamwater samples (dissolved = total) (fig. 4). About one-fifth of the AMD samples exceeded the Commonwealth of Pennsylvania (1998a, 1998b) criteria for instantaneous maximum concentration of manganese in effluent from an active mine (5.0 mg/L), and more than one-third of the streamwater samples exceeded the Commonwealth of Pennsylvania (2002) instream criteria for total recoverable manganese (1.0 mg/L) (fig. 4). All AMD and most stream samples exceeded the U.S. Environmental Protection Agency (2002b) secondary drinking-water standard for manganese (0.05 mg/L).

Aluminum concentrations in the low and high base-flow AMD samples ranged from less than 0.01 to 17 mg/L, and in streamwater samples ranged from less than 0.01 to 8.9 mg/L (fig. 3, table 3). Dissolved aluminum concentration was inversely correlated with pH; AMD and streamwater samples that had pH greater than 5 had aluminum concentrations less than 0.75 mg/L. With few exceptions, dissolved aluminum concentrations at a particular AMD site were similar during low and high base-flow conditions; however, at most streamwater

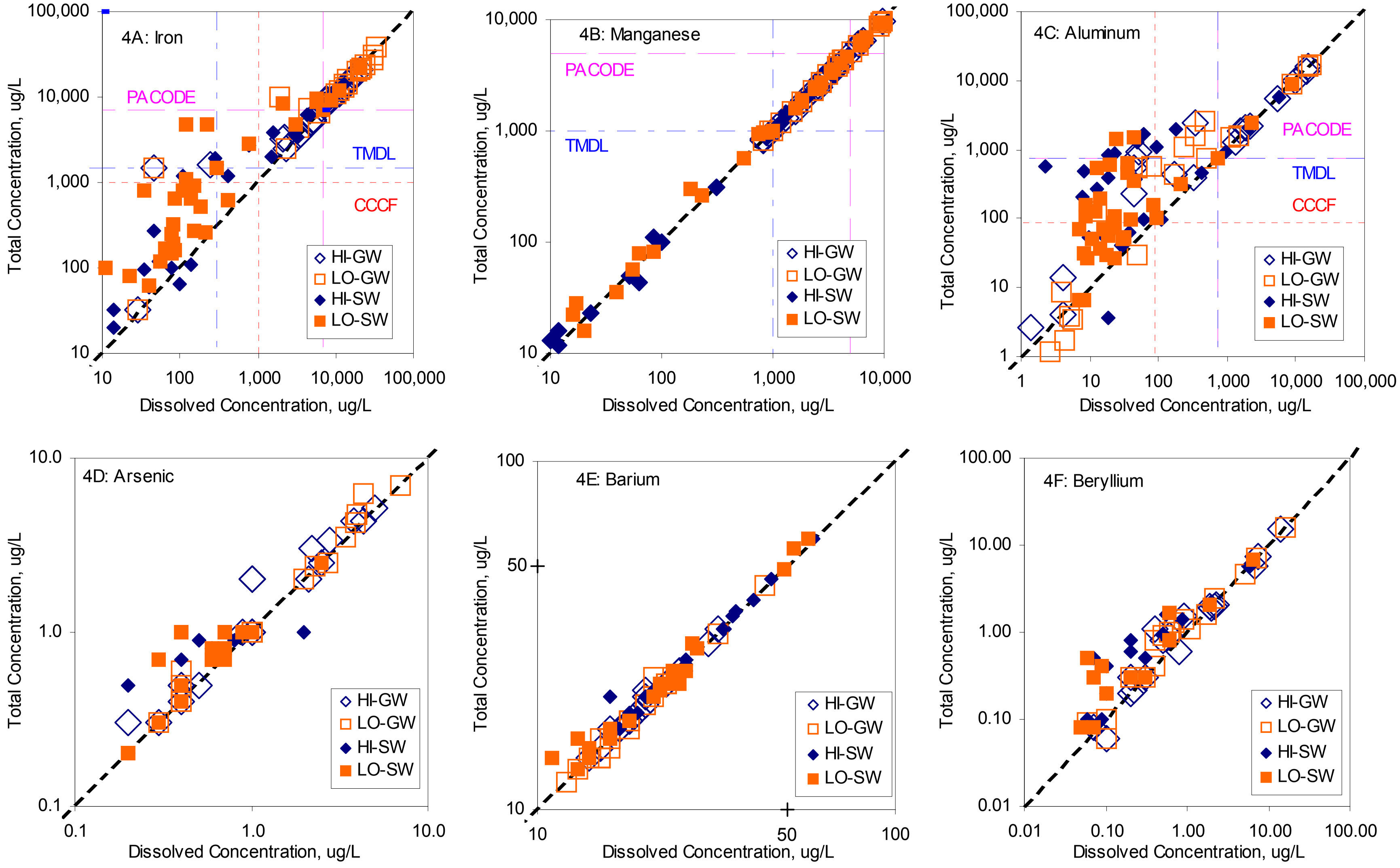

Figure 4. Comparison of total (whole-water) and dissolved (0.45-µm pore-size filter) concentrations, in micrograms per liter (µg/L), for low base-flow and high base-flow abandoned-mine drainage (HI-GW; LO-GW) and streamwater (HI-SW; LO-SW) samples collected in Mahanoy Creek Basin, Pennsylvania, March 2001 and August 2001, respectively: (A) iron, (B) manganese, (C) aluminum, (D) arsenic, (E) barium, (F) beryllium, (G) cadmium, (H) cobalt, (I) copper, (J) lead, (K) nickel, and (L) zinc. PACODE indicates effluent limitation for active surface or underground coal mine in Pennsylvania (Commonwealth of Pennsylvania, 1998a, b); TMDL indicates total maximum daily load stream-quality criteria (Commonwealth of Pennsylvania, 2001a); and CCCF indicates continuous contaminant concentration exposure tolerated by freshwater aquatic organisms without unacceptable effect (U.S. Environmental Protection Agency, 2002a).

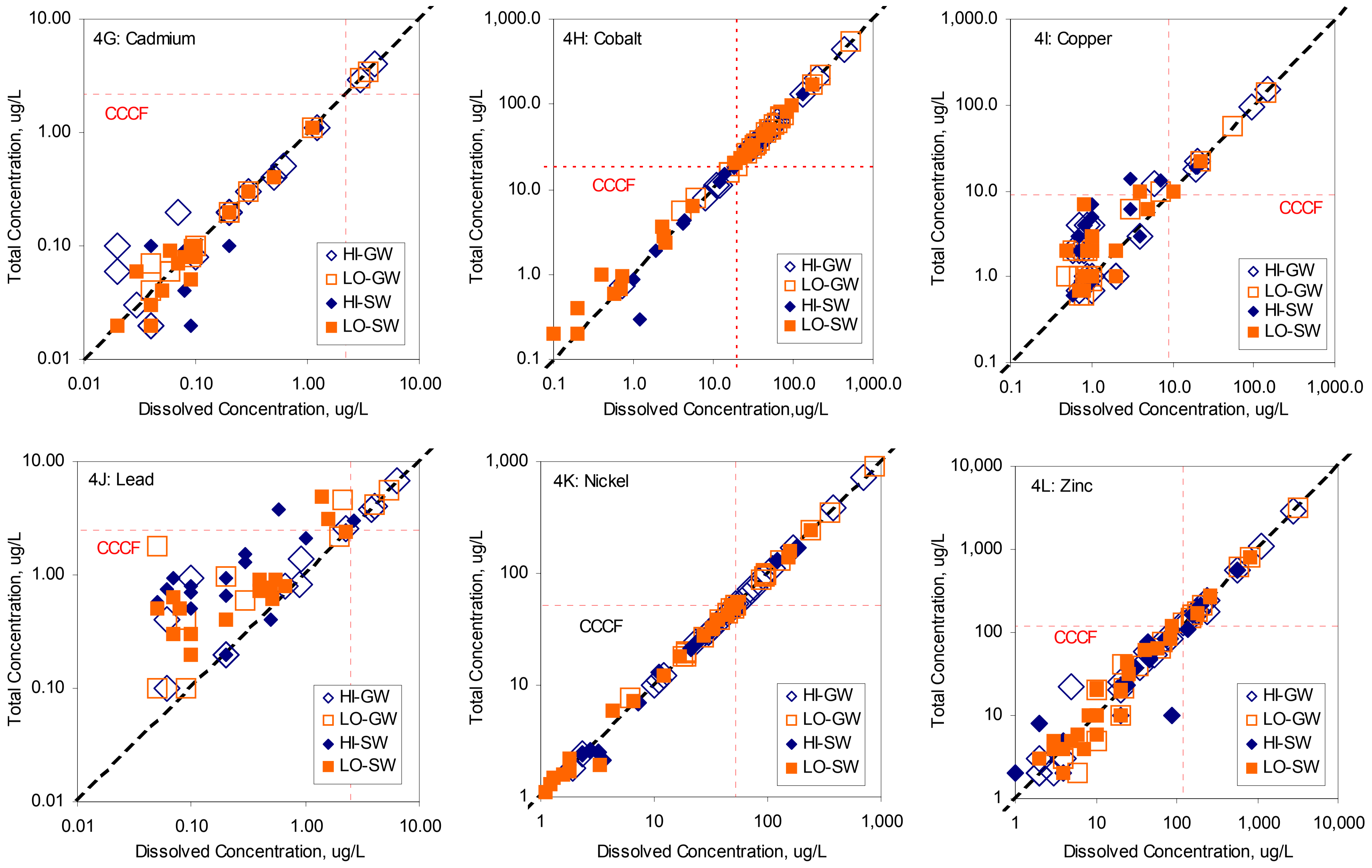

Figure 4. Comparison of total (whole-water) and dissolved (0.45-µm pore-size filter) concentrations, in micrograms per liter (µg/L), for low base-flow and high base-flow abandoned-mine drainage (HI-GW; LO-GW) and streamwater (HI-SW; LO-SW) samples collected in Mahanoy Creek Basin, Pennsylvania, March 2001 and August 2001, respectively: (A) iron, (B) manganese, (C) aluminum, (D) arsenic, (E) barium, (F) beryllium, (G) cadmium, (H) cobalt, (I) copper, (J) lead, (K) nickel, and (L) zinc. PACODE indicates effluent limitation for active surface or underground coal mine in Pennsylvania (Commonwealth of Pennsylvania, 1998a, b); TMDL indicates total maximum daily load stream-quality criteria (Commonwealth of Pennsylvania, 2001a); and CCCF indicates continuous contaminant concentration exposure tolerated by freshwater aquatic organisms without unacceptable effect (U.S. Environmental Protection Agency, 2002a).—Continued

sites, concentrations of aluminum differed between low and high base-flow conditions (fig. 3). For the AMD and streamwater samples containing more than 1.0 mg/L (1,000 µg/L) aluminum, dissolved aluminum was predominant (dissolved = total) (fig. 4). More than one-fourth of the AMD and one-tenth of the streamwater samples exceeded the Commonwealth of Pennsylvania (1998a, 1998b) criteria for instantaneous maximum concentration of aluminum in effluent from an active mine (0.75 mg/L) and the Commonwealth of Pennsylvania (2002) instream criteria for total recoverable aluminum (0.75 mg/L). Most AMD and approximately half the streamwater samples also exceeded continuous exposure criteria for protection of aquatic life from dissolved aluminum (0.087 mg/L) and the secondary drinking-water standard for aluminum (0.2 mg/L) (U.S. Environmental Protection Agency, 2002a, 2002b).

Arsenic was detected at concentrations greater than or equal to 0.2 µg/L in more than two-thirds of the AMD and streamwater samples; the maximum concentration was 7.0 µg/L (appendix A). Despite potential for its mobilization from coal and shale, dissolved arsenic was present at only low concentrations likely because of the adsorption of arsenate and arsenite ions by iron oxides (Cravotta and others, 2001). The total concentration of arsenic was as much as 2.5 times that of the dissolved concentration (fig. 4). None of the AMD or streamwater samples exceeded the 150 µg/L continuous exposure criteria for protection of aquatic life (U.S. Environmental Protection Agency, 2002a) or the 10 µg/L primary drinking-water standard for arsenic (U.S. Environmental Protection Agency, 2002b).

Barium was detected at concentrations greater than or equal to 10 µg/L in all of the AMD and streamwater samples; the maximum concentration was 75 µg/L (appendix A). Generally, the barium was present as a dissolved constituent (fig. 4), but at only low concentrations because of solubility control by barite ($BaSO_4$) in solutions containing sulfate (Nordstrom and Alpers, 1999). None of the AMD or streamwater samples exceeded continuous exposure criteria for protection of aquatic life (4,100 µg/L) or the primary drinking-water standard (2,000 µg/L) for barium (U.S. Environmental Protection Agency, 2002a, 2002b).

Beryllium was detected at concentrations greater than or equal to 0.05 µg/L in two-thirds of the AMD and streamwater samples; the maximum concentration was 16 µg/L (appendix A). The beryllium was predominantly dissolved at concentrations greater than or equal to 1 µg/L (fig. 4). Although criteria for beryllium have not been established for protection of aquatic life, some AMD or streamwater samples exceeded primary drinking-water standards (4 µg/L) for beryllium (U.S. Environmental Protection Agency, 2002a, 2002b).

Cadmium was detected at concentrations greater than or equal to 0.02 µg/L in four-fifths of the AMD and streamwater samples; the maximum concentration was 4.0 µg/L (appendix A). The cadmium was predominantly dissolved at concentrations greater than or equal to 0.3 µg/L (fig. 4). Some AMD samples exceeded continuous exposure criteria for protection of aquatic life (2 µg/L); however, none of the AMD or streamwater samples exceeded the primary drinking-water standard (5 µg/L) for cadmium (U.S. Environmental Protection Agency, 2002a, 2002b).

Chromium (not shown in fig. 4) was detected at concentrations greater than or equal to 1.0 µg/L in less than one-third of the AMD and streamwater samples; the maximum concentration was 9 µg/L (appendix A). None of the AMD or streamwater samples exceeded continuous exposure criteria for protection of aquatic life (19 µg/L) or the primary drinking-water standard (100 µg/L) for chromium (U.S. Environmental Protection Agency, 2002a, 2002b).

Cobalt was detected at concentrations greater than or equal to 0.02 µg/L in all of the AMD and all but one of the streamwater samples; the maximum concentration was 530 µg/L (appendix A). The cobalt was predominantly dissolved at concentrations greater than or equal to 1 µg/L (fig. 4). Most of the AMD and associated streamwater samples exceeded continuous exposure criteria for protection of aquatic life (19 µg/L); drinking-water standards have not been established for cobalt (U.S. Environmental Protection Agency, 2002a, 2002b).

Copper was detected at concentrations greater than or equal to 0.5 µg/L in seven-eighths of the AMD and streamwater samples; the maximum concentration was 150 µg/L (appendix A). The copper was predominantly dissolved at concentrations greater than or equal to 10 µg/L (fig. 4). Some AMD and associated streamwater samples exceeded continuous exposure criteria for protection of aquatic life (9 µg/L); none exceeded the primary drinking-water standard (1,300 µg/L) for copper (U.S. Environmental Protection Agency, 2002a, 2002b).

Lead was detected at concentrations greater than or equal to 0.05 µg/L in more than half the AMD and streamwater samples; the maximum concentration was 6.4 µg/L (appendix A). The lead was predominantly dissolved at concentrations greater than or equal to 2.5 µg/L (fig. 4). Some AMD and one streamwater samples exceeded continuous exposure criteria for protection of aquatic life (2.5 µg/L); none exceeded the primary drinking-water standard (15 µg/L) for lead (U.S. Environmental Protection Agency, 2002a, 2002b).

Nickel was detected at concentrations greater than or equal to 0.3 µg/L in all the AMD and streamwater samples; the maximum concentration was 890 µg/L (appendix A). The nickel was predominantly dissolved (fig. 4). Many of the AMD and some streamwater samples exceeded continuous exposure criteria for protection of aquatic life (52 µg/L); drinking-water standards have not been established for nickel (U.S. Environmental Protection Agency, 2002a, 2002b).

Selenium (not shown in fig. 4) was detected at concentrations greater than 0.2 µg/L in less than one-tenth of the AMD and streamwater samples; the maximum concentration was 0.4 µg/L (appendix A). None of the AMD or streamwater samples exceeded continuous exposure criteria for protection of aquatic life (5.0 µg/L) or the primary drinking-water standard (50 µg/L) for selenium (U.S. Environmental Protection Agency, 2002a, 2002b).

Silver was detected at concentrations greater than or equal to 0.01 µg/L in less than one-eighth of the AMD and streamwater samples; the maximum concentration was 0.9 µg/L (appendix A). None of the AMD or streamwater samples exceeded the secondary drinking-water standard (200 µg/L); criteria for protection of aquatic life have not been established for silver (U.S. Environmental Protection Agency, 2002a, 2002b).

Thallium was detected at concentrations greater than or equal to 0.05 µg/L in less than one-twentieth of the AMD and streamwater samples; the maximum concentration was 0.2 µg/L (appendix A). None of the AMD or streamwater samples exceeded continuous exposure criteria for protection of aquatic life (13 µg/L) or the primary drinking-water standard (2 µg/L) for thallium (U.S. Environmental Protection Agency, 2002a, 2002b).

Vanadium was detected at concentrations greater than or equal to 0.2 µg/L in less than one-fourth of the AMD and streamwater samples; the maximum concentration was 1.0 µg/L (appendix A). None of the AMD or streamwater samples exceeded continuous exposure criteria for protection of aquatic life (100 µg/L); drinking-water standards have not been established for vanadium(U.S. Environmental Protection Agency, 2002a, 2002b).

Zinc was detected at concentrations greater than or equal to 1.0 µg/L in all but one each of the AMD and streamwater samples; the maximum concentration was 3,200 µg/L (appendix A). The zinc was predominantly dissolved (fig. 4). Many AMD and some streamwater samples exceeded continuous exposure criteria for protection of aquatic life (120 µg/L); however, none exceeded the secondary drinking-water standard (5,000 µg/L) for zinc (U.S. Environmental Protection Agency, 2002a, 2002b).

In summary, a majority of the base-flow streamwater samples during the study met Commonwealth of Pennsylvania (2001b, 2002) water-quality standards for pH (6.0 to 9.0); however, few met criteria for acidity less than alkalinity (net alkalinity greater than or equal to 20 mg/L as $CaCO_3$) and concentrations of dissolved iron (0.3 mg/L) and total manganese (1.0 mg/L) (figs. 3 and 4). Iron, aluminum, and various trace elements, including cobalt, copper, lead, nickel, and zinc, were present in various AMD and associated streamwater samples at concentrations at which continuous exposure can not be tolerated by aquatic organisms without an unacceptable effect. Furthermore, in some samples, concentrations of sulfate, iron, manganese, aluminum, and (or) beryllium exceeded drinking-water standards. Other trace elements, including antimony, arsenic, barium, cadmium, chromium, selenium, silver, and thallium, did not exceed concentration criteria for protection of aquatic organisms or human health. The characteristics at individual streamwater sites and specific source(s) of contaminants are summarized in the next section.

Flow and Quality of Streams

The flow and quality of low and high base-flow samples for each of the primary stream monitoring locations are described below, in approximate downstream order, by the local site identification number. The dominant upstream AMD sources and selected water-quality data are described in tables 1, 2, and 3 and shown in figures 5 and 6. The flow and quality of all the high base-flow streamwater samples are shown in figure 5. The flow and quality of those samples along the main stem of Mahanoy Creek are shown in figure 6. The predominant trends along the length of Mahanoy Creek from its headwaters to Ashland (S16) are increasing flow rates coupled with increasing pH and concentrations of iron, manganese, and sulfate associated with “net alkaline” AMD loading in the vicinity of Ashland. Downstream from Ashland to its mouth near Kneass (S30), continued increases in flow rates are coupled with near-neutral pH. Within the stream, the AMD contaminant concentrations are attenuated by dilution, precipitation, and adsorption processes. These downstream trends are similar to those reported by Sanders & Thomas, Inc. (1975, fig. 17), except that concentrations of acidity and sulfate have decreased compared to the 1975 report; the downstream segments of Mahanoy Creek presently (2001) have excess alkalinity.

Site S03 is near the headwaters of Mahanoy Creek below the village of Buck Mountain, 1.2 mi (1.9 km) downstream from the Vulcan-Buck Mountain Mine Morris Tunnel (M01) and upstream from other AMD sources. In March 2001, streamwater within the relatively undisturbed, unstained channel of the sampled reach was flowing at 0.67 ft^3/s (1,140 L/min). The streamwater had pH of 4.0, net alkalinity of -11 mg/L, sulfate concentration of 34 mg/L, and relatively low concentrations (less than 1.0 mg/L) and cumulative loading of dissolved iron, aluminum, and manganese (0.8 Mg/yr) (table 3; fig. 5). The concentration of dissolved nitrate in Mahanoy Creek at S03 was 0.4 mg/L (table 3). In August 2001, the sampled reach was dry. These data indicate that the area sampled by S03 is affected by losses of flow because of streambed leakage to underground mines and, when flowing, has water quality that is marginally affected by intermittent AMD.

Site S04 is on Mahanoy Creek at the eastern limit of Mahanoy City, 0.1 mi (0.16 km) downstream from S03, and is dominated by discharge from the Vulcan-Buck Mountain Mine seepage (M02). In March 2001, streamwater within the ochre-stained channel of the sampled reach for S04 was flowing at 8.9 ft^3/s (15,100 L/min) with pH of 4.8, net alkalinity of -20 mg/L, and dissolved concentrations of sulfate of 105 mg/L, iron of 3.2 mg/L, aluminum of 2.0 mg/L, manganese of 1.1 mg/L, and nitrate of 0.4 mg/L (table 3; figs. 5 and 6). In August 2001, the sampled reach at S04 was dry; the upstream AMD sources at M01 and M02 were not flowing (table 3, fig. 5). The data for Mahanoy Creek at S04 indicate that the area is affected by losses of flow and by loading of acidity and metals from intermittent AMD sources.

Site S06 is the mouth of North Mahanoy Creek at the northern limit of Mahanoy City and is 2.5 mi (4.0 km) downstream from S05 near its headwaters. Water flowing from S05 to S06 travels through an extensive mine-scarred area, but no large mine discharges are present in the subbasin. In March 2001, streamflow at S06 was 8.7 ft^3/s (14,800 L/min) with pH

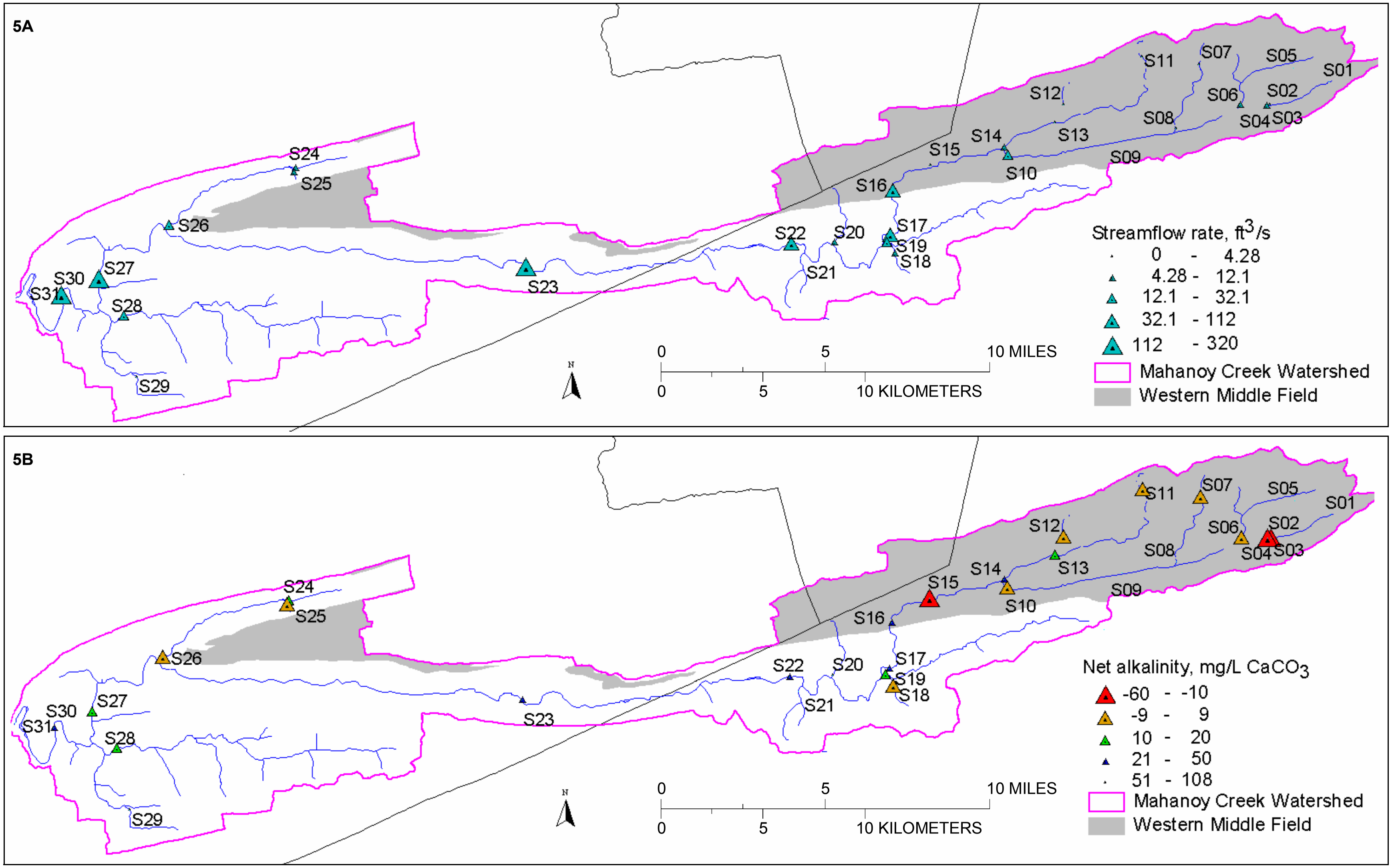

Figure 5. Mahanoy Creek Basin, Pennsylvania, showing: (A) streamflow; concentrations of (B) net alkalinity, (C) dissolved iron, (D) dissolved manganese, (E) dissolved aluminum, (F) sulfate, and (G) nitrate; and (H) Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity at streamwater sampling sites. Streamflow and chemical concentration data for March 2001; macroinvertebrate data for various dates, May to October, 2000-2002, at more numerous sites than sampled for water chemistry. Streamflow rate in cubic feet per second (ft^3/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L $CaCO_3$); and dissolved metals, sulfate, and nitrate concentrations in milligrams per liter (mg/L).

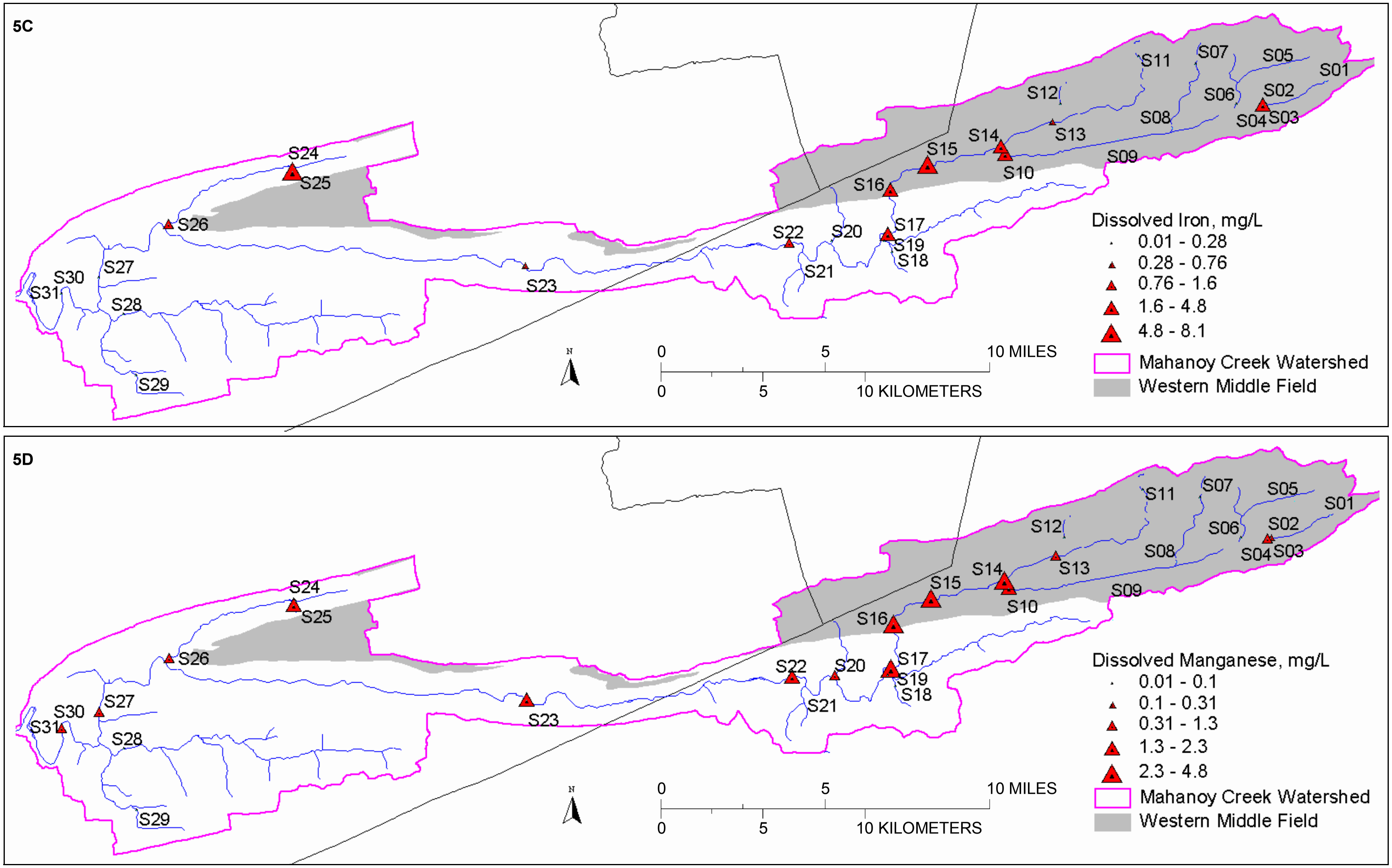

Figure 5. Mahanoy Creek Basin, Pennsylvania, showing: (A) streamflow; concentrations of (B) net alkalinity, (C) dissolved iron, (D) dissolved manganese, (E) dissolved aluminum, (F) sulfate, and (G) nitrate; and (H) Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity at streamwater sampling sites. Streamflow and chemical concentration data for March 2001; macroinvertebrate data for various dates, May to October, 2000-2002, at more numerous sites than sampled for water chemistry. Streamflow rate in cubic feet per second (ft^3/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L $CaCO_3$); and dissolved metals, sulfate, and nitrate concentrations in milligrams per liter (mg/L).—Continued

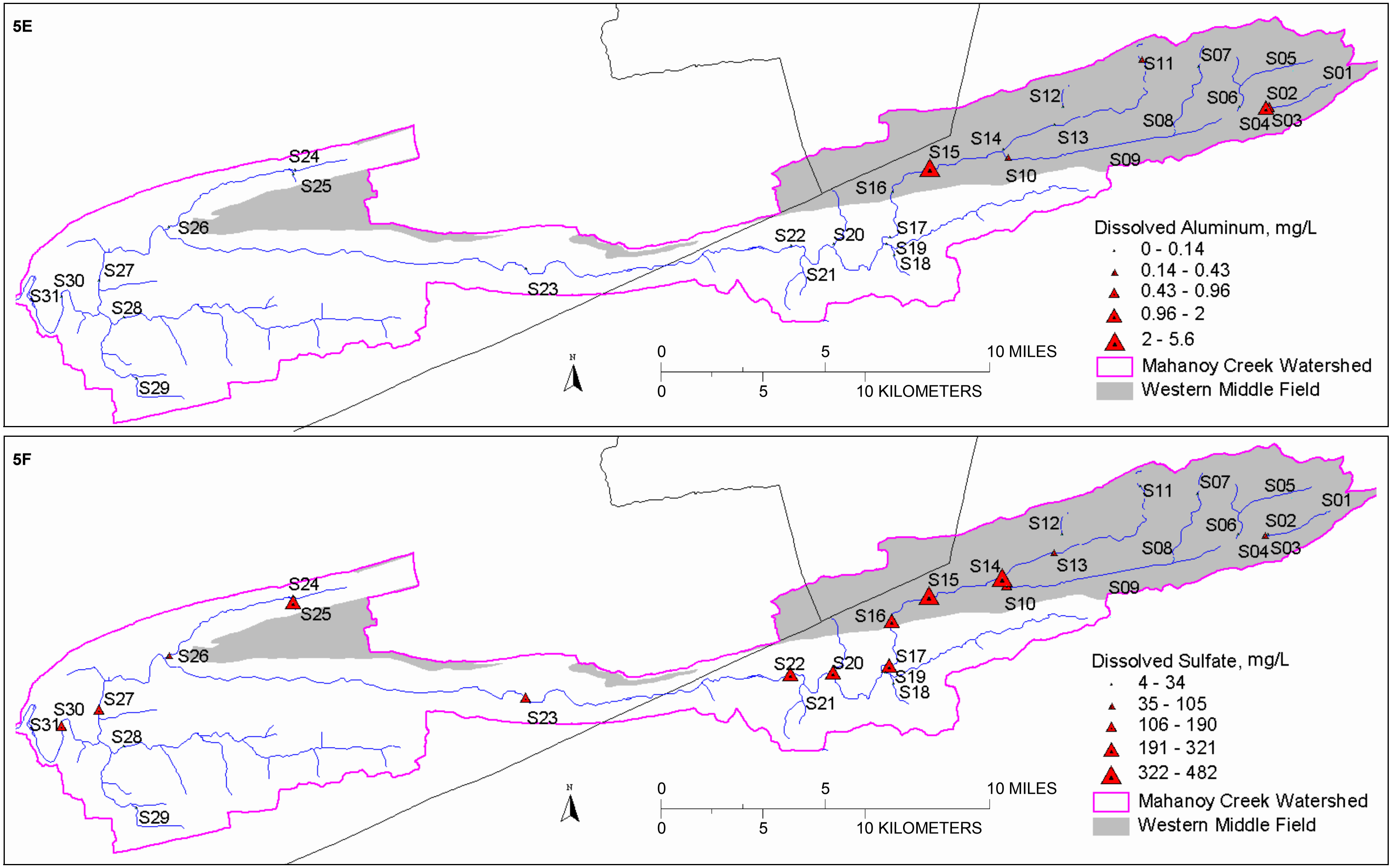

Figure 5. Mahanoy Creek Basin, Pennsylvania, showing: (A) streamflow; concentrations of (B) net alkalinity, (C) dissolved iron, (D) dissolved manganese, (E) dissolved aluminum, (F) sulfate, and (G) nitrate; and (H) Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity at streamwater sampling sites. Streamflow and chemical concentration data for March 2001; macroinvertebrate data for various dates, May to October, 2000-2002, at more numerous sites than sampled for water chemistry. Streamflow rate in cubic feet per second (ft^3/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L $CaCO_3$); and dissolved metals, sulfate, and nitrate concentrations in milligrams per liter (mg/L).—Continued

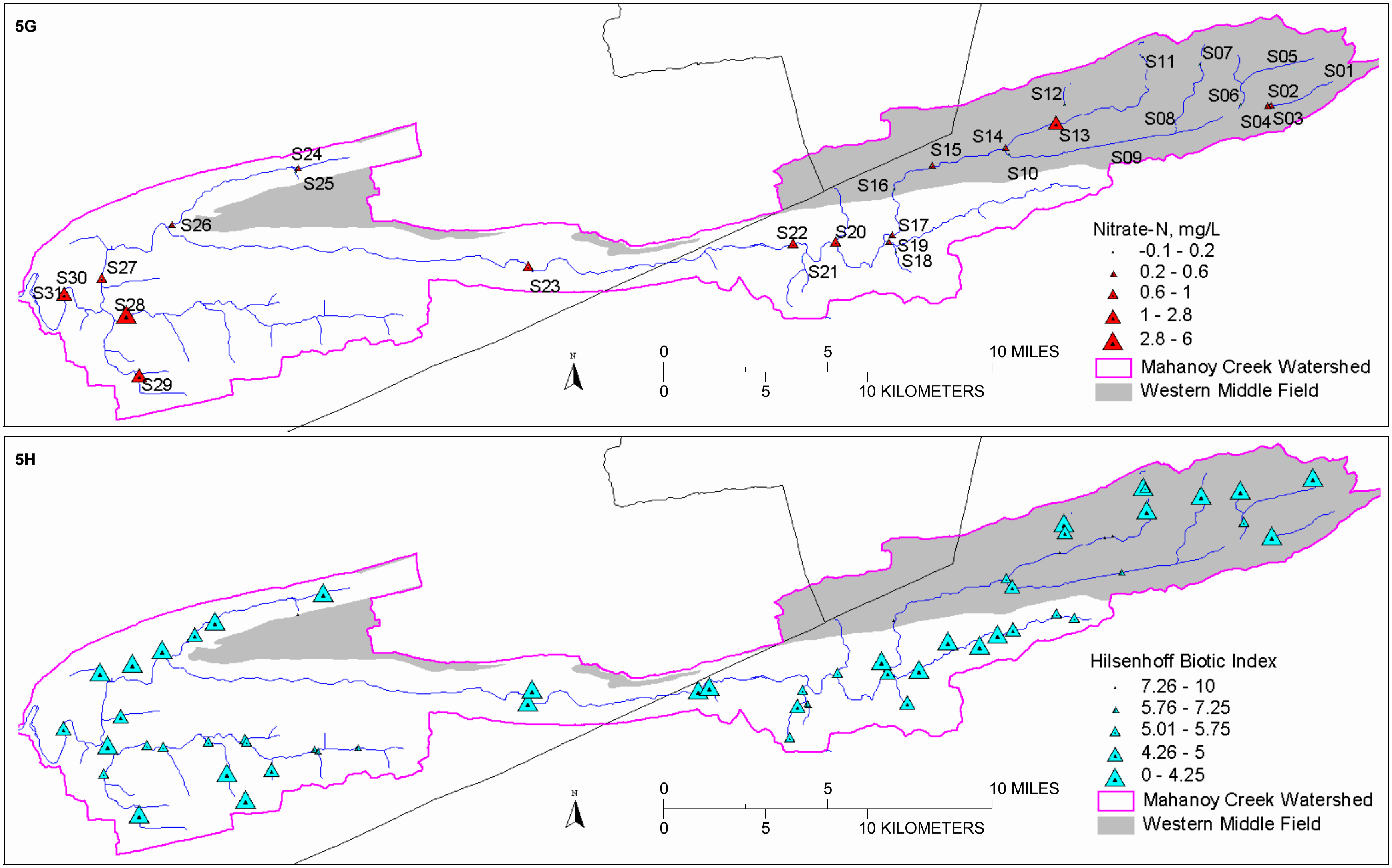

Figure 5. Mahanoy Creek Basin, Pennsylvania, showing: (A) streamflow; concentrations of (B) net alkalinity, (C) dissolved iron, (D) dissolved manganese, (E) dissolved aluminum, (F) sulfate, and (G) nitrate; and (H) Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity at streamwater sampling sites. Streamflow and chemical concentration data for March 2001; macroinvertebrate data for various dates, May to October, 2000-2002, at more numerous sites than sampled for water chemistry. Streamflow rate in cubic feet per second (ft^3/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L $CaCO_3$); and dissolved metals, sulfate, and nitrate concentrations in milligrams per liter (mg/L).—Continued

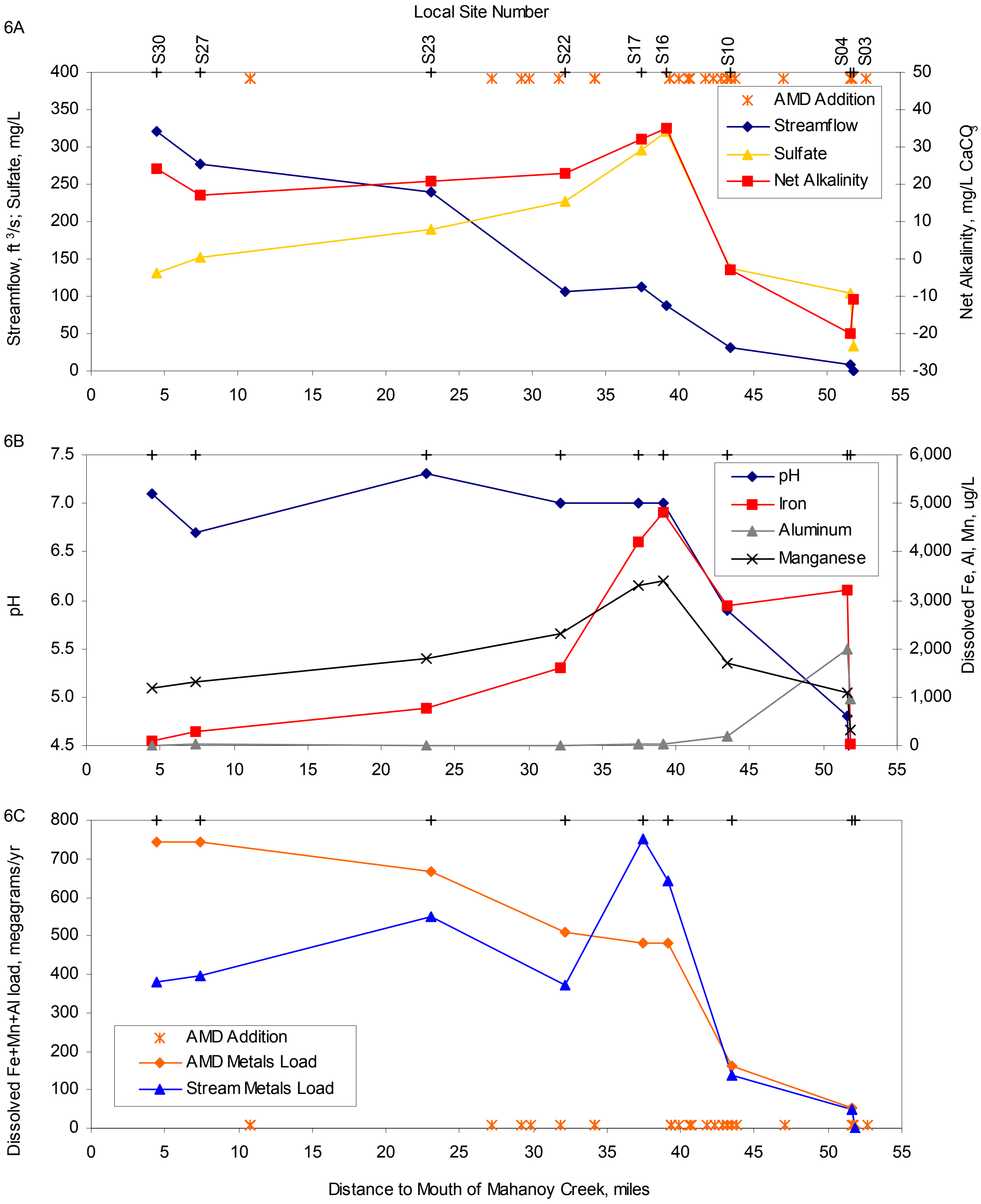

Figure 6. Streamflow and water quality at sites along the main stem of Mahanoy Creek, Pennsylvania, March 2001: (A) Streamflow and concentrations of sulfate and net alkalinity; (B) pH and concentrations of dissolved iron, aluminum, and manganese; and (C) loading of metals within stream and summed from upstream abandoned mine drainage (AMD) sources. Locations of AMD addition shown where the AMD discharge or recipient tributary enters Mahanoy Creek.

of 5.6, net alkalinity 1 mg/L, and concentrations of dissolved iron of 0.01 mg/L, aluminum of 0.06 mg/L, manganese of 0.02 mg/L, and nitrogen of 0.05 mg/L (table 3). In August 2001, despite measurable streamflow at site S05, the sampled reach at S06 was dry. These data indicate that the reach between sites S05 and S06 on North Mahanoy Creek is affected by losses of flow, but the water quality is not affected by AMD.

Site S08 is the mouth of Waste House Run at St. Nicholas and is 2.5 mi (4.0 km) downstream from S07. Streamwater at S07 originates in a forested "water-supply" area and then continues to S08 through a mine-scarred area, but no large mine discharges are present in the subbasin. In both March 2001 and August 2001, the reach at S08 was dry (table 3). Nevertheless, at S07, streamflows of 1.33 and 0.11 ft^3/s (2,260 and 187 L/min) in March 2001 and August 2001, respectively, were measured with near-neutral pH and low metals concentrations (table 3). These data indicate that the reach between S07 and S08 on Waste House Run is severely affected by losses of flow. The entire stream is captured by an open mine void characterized as a crop fall by Sanders & Thomas, Inc. (1975) approximately 0.3 mi (0.5 km) downstream of S07.

Site S09 is Mahanoy Creek at Gilberton, 4.5 mi (7.2 km) downstream from site S04, and includes flow from the Vulcan-Buck Mountain Boreholes discharge (M03) but is above the Gilberton Mine Pump (M04). Streamwater at S09 was sampled only in August 2001; streamflow was 2.8 ft^3/s (4,760 L/min), with pH of 5.0, net alkalinity of -6 mg/L, and dissolved concentrations of sulfate of 256 mg/L, iron of 0.41 mg/L, aluminum of 0.21 mg/L, manganese of 1.8 mg/L, nitrate of 2.0 mg/L, and phosphorus of 0.06 mg/L (table 3). These data indicate streamwater quality of Mahanoy Creek at S09 is affected by AMD and nutrient loading from Mahanoy City and vicinity.

Site S10 is Mahanoy Creek above Girardville, 3.5 mi (5.6 km) downstream from site S09, and includes intermittent flow from the Gilberton Mine Pump (M04) and continuous flow from the Girard Mine seepage (M11). In March and August 2001, flows at S10 were 32.1 and 7.84 ft^3/s (54,500 and 13,300 L/min), respectively (table 3). The quality of the high and low base-flow samples was variable with pH of 5.9 and 6.3, net alkalinity of -3 and 9 mg/L, and dissolved concentrations of sulfate of 138 and 203 mg/L, iron of 2.9 and 6.8 mg/L, aluminum of 0.18 and 0.01 mg/L, manganese of 1.7 and 2.6 mg/L, nitrate of 0.11 and 0.66 mg/L, and phosphorus of 0.01 and less than 0.01 mg/L (table 3; figs. 5 and 6). During both sampling events, the Gilberton Mine Pump was not operating and M04 was dry. The AMD discharged from the Girard Mine seepage at M11 accounted for one-eighth and one-third of the streamflow volume at S10 during March and August, respectively.

Site S13 is Shenandoah Creek at Lost Creek, downstream of streamwater sites on Kehly Run (S11) and Lost Creek (S12) and the borough of Shenandoah (fig. 1) plus nearby smaller municipalities (not shown on figure 1) including Shenandoah Heights, East William Penn, West William Penn, and Lost Creek. In March and August 2001, flows at S13 were 3.0 and 0.2 ft^3/s (5,098 and 340 L/min), respectively (table 3). The high and low base-flow samples had pH of 7.4 and 7.0, net alkalinity of 13 and 26 mg/L, and dissolved concentrations of sulfate of 74 and 166 mg/L, iron of 0.41 and 0.09 mg/L, aluminum of 0.05 and 0.01 mg/L, manganese of 0.87 and 1.8 mg/L, nitrate of 2.0 and 3.3 mg/L, and phosphorus of 0.30 and 0.33 mg/L (table 3, fig. 5). These data indicate streamwater quality of Shenandoah Creek at S13 is affected by AMD and nutrient loading. The nutrient enrichment is associated with treated sewage discharges from the city of Shenandoah and untreated sewage from the boroughs of East William Penn, West William Penn, and Lost Creek (M. A. Friday, Pennsylvania Department of Environmental Protection, written commun., 2003).

Site S14 is the mouth of Shenandoah Creek at Girardville, 1.8 mi (2.9 km) downstream from site S13 on Shenandoah Creek, and includes AMD discharges from the Weston Mine (M05, M06, M07) and the Hammond Mine (M08, M09, M10). The streamwater from S13 and Hammond Mine AMD discharges (M08-M10) flow into a large constructed wetland (exceeding 10 acres or 40,470 m^2) immediately upstream of S14. In March and August 2001, flows at S14 were 12.1 and 5.5 ft^3/s (20,600 and 9,340 L/min), respectively (table 3). The high and low base-flow samples had pH of 6.7 and 6.9, net alkalinity of 43 and 68 mg/L, and dissolved concentrations of sulfate of 421 and 553 mg/L, iron of 3.1 and 0.76 mg/L, aluminum of 0.01 mg/L, manganese of 4.0 and 6.9 mg/L, nitrate of 0.60 and 0.27 mg/L, and phosphorus of 0.03 mg/L (table 3, fig. 5). Larger streamflow, higher concentrations of acidity, sulfate, and metals, and lower concentrations of nitrogen and phosphorus at S14 than at S13 (table 3, fig. 5) indicate streamwater quality of Shenandoah Creek at S14 is affected by large discharges of AMD downstream from S13. The nutrient enrichment at S13 is largely attenuated by mixing with AMD and biological uptake and other processes within the wetland upstream of S14.

Site S15 is at the mouth of the unnamed tributary locally called Big Mine Run, 0.5 mi (0.8 km) downstream from the Centralia Mine Tunnel discharge (M19). Big Mine Run is formed almost entirely of AMD discharged from the Centralia Mine Tunnel; the flow rate was assumed the same for M19 and S15 and was measured at an intermediate location. In March and August 2001, flows at S15 were 3.86 and 2.43 ft^3/s (6,560 and 4,130 L/min), respectively (table 3). The high and low base-flow samples at S15 had pH of 3.7 and 3.6, net alkalinity of -60 and -79 mg/L, and dissolved concentrations of sulfate of 482 and 524 mg/L, iron of 5.8 and 3.1 mg/L, aluminum of 5.6 and 8.9 mg/L, manganese of 4.8 and 6.0 mg/L, nitrate of 0.50 and less than 0.10 mg/L, and phosphorus of less than 0.01 mg/L (table 3, fig. 5). Site S15 had the most severely affected streamwater quality in the Mahanoy Creek Basin on the basis of net alkalinity (acidity) and dissolved metals concentrations, particularly aluminum.

Site S16 is Mahanoy Creek downstream from Ashland and Girardville, 4.3 mi (6.9 km) downstream from site S10, and includes Shenandoah Creek and various large AMD discharges (M12-M25). Site S16 is near the downstream limit of the Western Middle Anthracite Field that underlies the upper Mahanoy Creek Basin (fig. 5). The AMD sources include the Packer #5 Borehole and Breach (M12, M13, M14), the Preston Mine #2 and #3 discharges (M15, M16, M17), the Bast Mine Tunnel (M18), the Centralia Mine Tunnel (M19), the Bast Mine overflow and Oakland Tunnel (M20, M21), the Tunnel Mine drain pool area (M22, M23, M24), and the Tunnel Mine Orchard Drift (M25). In March and August 2001, flows at S16 were 87.7 and 50.2 ft^3/s (149,000 and 85,300 L/min), respectively (table 3). The high and low base-flow samples at S16 had pH of 7.0, net alkalinity of 35 and 42 mg/L, and dissolved concentrations of sulfate of 321 and 491 mg/L, iron of 4.8 and 5.7 mg/L, aluminum of 0.02 mg/L, manganese of 3.4 and 6.5 mg/L, nitrate of 0.16 and 0.30 mg/L, and phosphorus of 0.03 and 0.04 mg/L (table 3; figs. 5 and 6). The metals loading of 548 Mg/yr (604 ton/yr) at S15 during August 2001 was larger than that at any other site (table 3), whereas the metals loading of 644 Mg/yr (710 ton/yr) during March 2001 was exceeded (or equalled) only by that at the next site downstream (S17).

Site S17 is Mahanoy Creek at Gordon, 1.7 mi (2.7 km) downstream from site S16. Although seepage from coal silt and wetlands along the floodplain indicate potential for AMD loading, no large AMD sources and only one unnamed tributary from the unmined area of Fountain Springs are present between S17 and S16. In March and August 2001, flows at S17 were ~112 (error?) and 46.2 ft^3/s (190,000 and 78,500 L/min), respectively (table 3). The high and low base-flow samples at S17 had pH of 7.0 and 6.9, net alkalinity of 32 and 51 mg/L, and dissolved concentrations of sulfate of 295 and 480 mg/L, iron of 4.2 and 2.1 mg/L, aluminum of 0.02 and 0.04 mg/L, manganese of 3.3 and 6.0 mg/L, nitrate of 0.30 and 0.45 mg/L, and phosphorus of 0.04 mg/L (table 3; figs. 5 and 6). Generally, the metals concentrations and loading were expected to be lower at S17 than S16 because of negligible additional AMD sources and attenuation of dissolved constituents by dilution and mineral precipitation within the streambed. In March 2001, concentrations of sulfate, iron, and manganese declined by 8, 13, and 3 percent, respectively, at S17 compared to S16. However, the loading and flow rate at S17 were 17 and 28 percent larger, respectively, than those at S16 in March 2001, and also were larger than those for a downstream site, S22, indicating a possible positive error in the flow and associated loadings values for S17. If the flow at S17 in March 2001 was 95.3 ft^3/s (161,900 L/min), consistent with the volume increase to dilute sulfate from the observed concentration at S16 to that at S17, the metals loading at S17 would be 641 Mg/yr (707 ton/yr) and comparable to that at S16.

Site S19 is the mouth of Little Mahanoy Creek at Gordon, downstream of Rattling Run (S18). No mining has taken place in the Little Mahanoy Creek subbasin (Sanders & Thomas, Inc., 1975). The borough of Frackville is at the headwaters of Little Mahanoy Creek. In March and August 2001, flows at S19 were 30.1 and 6.8 ft^3/s (51,100 and 11,600 L/min), respectively (table 3). The high and low base-flow samples at S19 had pH of 7.0, net alkalinity of 11 and 20 mg/L, and dissolved concentrations of sulfate of 10 and 11 mg/L, iron of 0.10 and 0.08 mg/L, aluminum of 0.11 and 0.01 mg/L, manganese of 0.05 and 0.02 mg/L, nitrate of 0.55 and 0.52 mg/L, and phosphorus of 0.03 and 0.07 mg/L (table 3, fig. 5). Similar water-quality results, but with slightly lower pH and nutrient levels, were obtained for site S18 near the mouth of Rattling Run (table 3, fig. 5). These data are consistent with the expectation that mining has not affected the quality of Little Mahanoy Creek; however, nutrient loading affects its water quality.

Site S20 is at the mouth of the unnamed tributary locally called Big Run, approximately 1.9 mi (3.1 km) downstream from the Potts Mine discharges (M26 and M27) near Locustdale. In March and August 2001, flows at S20 were 6.08 and 1.33 ft^3/s (10,300 and 2,260 L/min), respectively (table 3). The high and low base-flow samples at S20 had pH of 8.2 and 8.0, net alkalinity of 108 and 178 mg/L, and dissolved concentrations of sulfate of 286 and 508 mg/L, iron of 0.12 and 0.22 mg/L, aluminum of 0.03 and 0.01 mg/L, manganese of 0.90 and 0.75 mg/L, nitrate of 0.80 and 0.50 mg/L, and phosphorus of less than 0.01 mg/L (table 3, fig. 5). Although the flow and quality of Big Run are dominated by iron-laden, net alkaline AMD discharged from the Potts Mine (M26 and M27), the large excess net alkalinity and turbulent flow path facilitated the oxidation and attenuation of iron and associated metals within the streambed between the AMD sources and the mouth of Big Run.

Site S21 is near the mouth of Crab Run. Streamwater at S21 was sampled only in August 2001; streamflow was 0.61 ft^3/s (1,040 L/min), with pH of 7.1, net alkalinity of 32 mg/L, and dissolved concentrations of sulfate of 9 mg/L, iron of 0.15 mg/L, aluminum of 0.01 mg/L, manganese of 0.05 mg/L, nitrate of 3.3 mg/L, and phosphorus 0.08 mg/L (table 3). Although Crab Run was initially listed by PaDEP as degraded because of AMD (Pennsylvania Department of Environmental Protection, 1998), no mining has taken place in the Crab Run subbasin, and AMD constituents are not elevated. Fine silt covered the cobble-dominated streambed. Agricultural activities in the subbasin are possible sources of nutrients and other water-quality stresses.

Site S22 is Mahanoy Creek near Lavelle, 5.1 mi (8.2 km) downstream from S17, and includes Little Mahanoy Creek, Big Run, and Crab Run. In March and August 2001, flows at S22 were 107 and 51.4 ft^3/s (182,000 and 87,300 L/min), respectively (table 3). The high and low base-flow samples at S22 had pH of 7.0 and 7.6, net alkalinity of 23 and 47 mg/L, and dissolved concentrations of sulfate of 227 and 427 mg/L, iron of 1.6 and 0.12 mg/L, aluminum of less than 0.01 and 0.04 mg/L, manganese of 2.3 and 4.6 mg/L, nitrate of 0.70 and 0.76 mg/L, and phosphorus of 0.03 and 0.04 mg/L (table 3; figs. 5 and 6). The metals loadings at S22 of 373 Mg/yr (411 ton/yr) and 219 Mg/yr (241 ton/yr) during March 2001 and August 2001, respectively, were about half that at Mahanoy Creek at Ashland (S16) or Gordon (S17) for the same time period. These data indicate that dissolved iron and aluminum concentrations from AMD sources (M1-M27) largely have been attenuated by mineral precipitation within the streambed upstream from S22 because of the near-neutral pH and excess alkalinity of Mahanoy Creek.

Site S23 is Mahanoy Creek near Gowen City, 9.0 mi (14.5 km) downstream from S22, and includes large AMD discharges from the Locust Gap Tunnel (M29; called the Helfenstein Tunnel by Sanders & Thomas, Inc., 1975) and the Doutyville Tunnel (M31) that divert water from underground mines in the Locust Gap area of the Shamokin Creek Basin to the north. In March and August 2001, flows at S23 were 239 and 65.5 ft^3/s (406,000 and 111,000 L/min), respectively (table 3). The high and low base-flow samples at S23 had pH of 7.3 and 7.6, net alkalinity of 21 and 58 mg/L, and dissolved concentrations of sulfate of 190 and 409 mg/L, iron of 0.76 and 0.11 mg/L, aluminum of 0.01 and 0.02 mg/L, manganese of 1.8 and 3.3 mg/L, nitrate of 1.0 and 0.73 mg/L, and phosphorus of 0.03 and less than 0.01 mg/L (table 3; figs. 5 and 6). The metals loadings at S23 of 549 Mg/yr (604 ton/yr) and 201 Mg/yr (221 ton/yr) during March 2001 and August 2001, respectively, were greater during March but comparable during August to that at Mahanoy Creek near Lavelle (S22).

Site S24 is on Zerbe Run below Trevorton and is not affected by mining (table 3, fig. 5). In March and August 2001, flows at S24 were 7.99 and 0.62 ft^3/s (13,600 and 1,050 L/min), respectively (table 3). The high and low base-flow samples at S24 had pH of 7.0 and 7.2, net alkalinity of 11 and 49 mg/L, and dissolved concentrations of sulfate of 10 and 8 mg/L, iron of 0.14 and 0.21 mg/L, aluminum of 0.04 and 0.02 mg/L, manganese of 0.06 and 0.02 mg/L, nitrate of 0.51 and 0.72 mg/L, and phosphorus of 0.01 mg/L (table 3, fig. 5).

Site S25 is near the mouth of an unnamed tributary to Zerbe Run at Trevorton, 0.7 mi (1.1 km) downstream from the North Franklin Mine AMD discharges (M32, M33). In March and August 2001, flows at S25 were 6.68 and 2.79 ft^3/s (11,300 and 4,740 L/min), respectively (table 3). The North Franklin Mine Drift and Borehole (M32) accounted for most of this streamflow (table 3). The high and low base-flow samples at S25 had pH of 6.3, net alkalinity of -6 mg/L, and dissolved concentrations of sulfate of 253 and 304 mg/L, iron of 8.1 and 11 mg/L, aluminum of 0.05 and 0.04 mg/L, manganese of 2.0 and 2.4 mg/L, nitrate of 0.07 and less than 0.01 mg/L, and phosphorus of less than 0.01 mg/L (table 3, fig. 5).

Site S26 is at the mouth of Zerbe Run, 4.6 mi (7.4 km) downstream from the junction of Zerbe Run at Trevorton (S24) and the unnamed tributary draining the North Franklin Mine (S25), and downstream from additional seepage from the North Franklin Mine. In March and August 2001, flows at S26 were 30.2 and 4.36 ft^3/s (51,300 and 7,400 L/min), respectively (table 3). During low base flow, the unnamed tributary draining the North Franklin Mine (S25) accounted for almost two-thirds of this streamflow; however, during high base flow, contributions from S24 and S25 were similar and, combined, accounted for only half the total streamflow at S26 (table 3). The quality of the high and low base-flow samples at S26 was variable with pH of 6.0 and 4.3, net alkalinity of -3 and -20 mg/L, and dissolved concentrations of sulfate of 99 and 264 mg/L, iron of 1.5 and 0.15 mg/L, aluminum of 0.10 and 2.3 mg/L, manganese of 0.93 and 2.7 mg/L, nitrate of 0.50 and 0.60 mg/L, and phosphorus of 0.04 and 0.02 mg/L (table 3, fig. 5). Relatively poor water quality during low base-flow conditions reflects the predominance of AMD contributions; however, improved water quality during high base flow indicates substantial dilution by various unidentified sources of net-alkaline water quality.

Site S27 is Mahanoy Creek at Dornsife, 3.6 mi (5.8 km) downstream from S26 and 15.5 mi (24.9 km) downstream from S23. Downstream from its confluence with Zerbe Run at S26, no additional AMD sources contribute flow to Mahanoy Creek. In March and August 2001, flows at S27 were 277 and 71.2 ft^3/s (471,000 and 121,000 L/min), respectively (table 3). The high and low base-flow samples at S27 had pH of 6.7 and 6.9, net alkalinity of 17 and 38 mg/L, and dissolved concentrations of sulfate of 152 and 382 mg/L, iron of 0.28 and 0.08 mg/L, aluminum of 0.02 and 0.04 mg/L, manganese of 1.3 and 0.90 mg/L, nitrate of 0.66 and 0.80 mg/L, and phosphorus of 0.02 and less than 0.01 mg/L (table 3; figs. 5 and 6). The metals loadings at S27 were 396 Mg/yr (437 ton/yr) and 65.1 Mg/yr (71.7 ton/yr) during March 2001 and August 2001, respectively.

Site S28 is near the mouth of Schwaben Creek near Red Cross. Agriculture is the predominant land use in this 22.7 mi^2 (58.8 km^2) subbasin; coal-bearing rocks and associated discharges are absent in the subbasin. In March and August 2001, flows at S28 were 28.4 and 1.66 ft^3/s (48,300 and 2,820 L/min), respectively (table 3). The high and low base-flow samples at S28 had pH of 7.3 and 7.5, net alkalinity of 14 and 54 mg/L, and dissolved concentrations of sulfate of 12 and 11 mg/L, iron of 0.03 and 0.06 mg/L, aluminum of 0.01 and 0.02 mg/L, manganese of 0.01 and 0.02 mg/L, nitrate of 6.0 and 1.9 mg/L, and phosphorus of 0.01 and 0.03 mg/L (table 3, fig. 5). These data indicate constituents associated with AMD are not elevated; however, nitrogen enrichment is consistent with base-flow data collected in June 1993 by USGS (Durlin and Schaffstall, 1994, p. 273) indicating water-quality impairment from agricultural sources in the subbasin.

Site S29 is near the headwaters of Mouse Creek below Urban. Mouse Creek ultimately flows to Schwaben Creek; the mouth of Mouse Creek is 2.6 mi (4.2 km) downstream from S29 and 0.5 mi (0.8 km) downstream from S28. Approximately half the 2.8 mi^2 (7.3 km^2) subbasin above S29 is covered by agricultural fields and half is forested. In March and August 2001, flows at S29 were 4.28 and 0.52 ft^3/s (7,270 and 883 L/min), respectively (table 3). The high and low base-flow samples at S29 had pH of 8.4 and 7.7, net alkalinity of 52 and 120 mg/L, and dissolved concentrations of sulfate of 16 and 17 mg/L, iron of 0.05 and 0.06 mg/L, aluminum of 0.02 and 0.01 mg/L, manganese of 0.01 mg/L, nitrate of 2.8 mg/L, and phosphorus of 0.01 and 0.02 mg/L (table 3, fig. 5). Agricultural sources contribute nitrogen at site S29, which otherwise has exceptionally good water quality, consistent with base-flow data collected in June 1993 by USGS (Durlin and Schaffstall, 1994).

Site S30 is Mahanoy Creek at Kneass, 3.0 mi (4.8 km) downstream from S27 and 1.9 mi (3.0 km) downstream from the mouth of Schwaben Creek. In March and August 2001, flows at S30 were 320 and 76.6 ft^3/s (544,000 and 130,000 L/min), respectively (table 3). The high and low base-flow samples at S30 had pH of 7.1 and 7.2, net alkalinity of 24 and 21 mg/L, and dissolved concentrations of sulfate of 132 and 363 mg/L, iron of 0.11 and 0.08 mg/L, aluminum of 0.01 and 0.03 mg/L, manganese of 1.2 and 0.54 mg/L, nitrate of 1.8 and 0.77 mg/L, and phosphorus of 0.01 and less than 0.01 mg/L (table 3, fig. 5). The metals loadings at S30 were 378 Mg/yr (417 ton/yr) and 44.7 Mg/yr (49.3 ton/yr) during March 2001 and August 2001, respectively.

Site S31 is Mahanoy Creek near Herndon, 3.9 mi (6.3 km) downstream from S30. Streamwater at S31 was sampled only in August 2001; streamflow was 76.3 ft^3/s (130,000 L/min), with pH of 7.3, net alkalinity of 43 mg/L, and dissolved concentrations of sulfate of 361 mg/L, iron of 0.08 mg/L, aluminum of 0.02 mg/L, manganese of 0.23 mg/L, nitrate of 0.71 mg/L, and phosphorus of less than 0.01 mg/L (table 3). During August 2001, the metals loading at S31 was 22.2 Mg/yr (24.4 ton/yr), which was half that at S30. The decreased concentration of dissolved manganese from S30 to S31 accounted for the corresponding reduction in metals loading. The data for dissolved and total metals concentrations at S31 are consistent with base-flow data collected in September 1994 by USGS (Breen and Gavin, 1995) indicating transport of iron and aluminum is largely as suspended colloids, whereas manganese is largely dissolved (particle size less than 0.45 μm).

Streamflow Variability

Continuous data are not available to indicate the ranges in streamflow or water quality of Mahanoy Creek. However, long-term (1940-1992), continuous records are available for streamflow of Shamokin Creek near Shamokin (USGS station 01554500). Although continuous streamflow gaging was discontinued by USGS at this site in 1992, measurements were resumed during a recent study (Cravotta and Kirby, 2004a). The average daily streamflow at the Shamokin station during March 26-28, 2001, was 137 ft^3/s (232,800 L/min) and during August 21-23, 2001, was 30 ft^3/s (50,970 L/min), bracketing the long-term average streamflow of 85.1 ft^3/s (144,600 L/min) during 1940-1992. Hence, considering these data as a surrogate for conditions on Mahanoy Creek, the hydrological data collected for the high and low base-flow surveys of the Mahanoy Creek Basin likely bracket the long-term average conditions.

Wide ranges in flow rates at water-quality sampling sites were recorded during March and August 2001 (fig. 3) reflecting expected seasonal variability in recharge and ground-water discharge (base flow) in the Mahanoy Creek Basin. These interactions were amplified by drought conditions during the study period. During August 2001, 7 of the 29 surveyed AMD sources were “dry,” including the Vulcan-Buck Mountain Mine seepage (M02) and the Bast Mine overflow (M20) that were the seventh and eleventh largest AMD sources sampled during March 2001, respectively (tables 1 and 3). Six stream survey sites, on Mahanoy Creek near its headwaters (S01, S02, S03, S04) above Mahanoy City, North Mahanoy Creek at its mouth (S06) at Mahanoy City, and Waste House Run at St. Nicholas (S08), were dry in August 2001 (tables 2 and 3). In contrast, during the high base-flow survey in March 2001, only 1 of the 31 stream survey sites, Waste House Run at St. Nicholas (S08), and only 2 of the 29 sampled AMD sources, the Vulcan-Buck Mtn. Mine Morris Tunnel (M01) and Gilberton Mine Pump (M04) discharges, were not flowing (table 3). Of the 31 AMD sites reported from the Scarlift investigation, only 1 had been identified previously as intermittently flowing (Sanders & Thomas, Inc., 1975).

To determine the magnitude and effects of interbasin ground-water or surface-water diversions, streamflow at points along Mahanoy Creek and its tributaries was compared to the drainage area defined by upstream topography. Streamflow values were normalized as the yield, or discharge divided by the contributing area (figs. 7A and 7B). Streamflow yields for Mahanoy Creek at Kneass (S30) were larger during the high base-flow survey and smaller during the low base-flow survey than those for Mahanoy Creek near Ashland (S16). This result indicates AMD is a substantial, sustained source of base flow in the upper basin and is consistent with interpretations by Becher (1991) for the adjacent Shamokin Creek Basin. Becher noted that streamflow in the upper Shamokin Creek Basin is sustained by discharges from water stored in the mines and, consequently, is less variable than that for nearby unmined basins with equivalent watershed areas but greater proportions of runoff contributing to the streamflow. A more detailed analysis of the annual water budget for the Mahanoy Creek Basin and various subbasins is given by Sanders & Thomas, Inc. (1975).

Compared with Little Mahanoy Creek at Gordon (S19; unmined subbasin), Mahanoy Creek at Ashland (S16) and Gordon (S17) had high streamflow yields for low base-flow conditions (fig. 7B). This difference also is apparent, but is less pronounced, during low base flow for Mahanoy Creek at Gowen City (S23) below large discharges from the Locust Gap Mine (M29, M31) that divert flow from the adjoining Shamokin Creek Basin. As explained by Sanders & Thomas, Inc. (1975), compared to unmined areas, greater yields from mined, unreclaimed, poorly vegetated areas result from a smaller proportion of the annual rainfall leaving the subbasin as evapotranspiration and a greater proportion recharged to ground water. In parts of the basin, surface flow is captured by subsidence pits, fractures, and other openings to the underground mines that ultimately discharge AMD at downflow locations. The diversion of water from the Shamokin Creek Basin or other subbasins magnifies this effect. Yield generally declined downstream from Gowen City indicating smaller recharge rates and potentially smaller effects from AMD. In contrast, some other tributaries had substantially smaller yields than Little Mahanoy Creek at Gordon, including upper Mahanoy Creek sites (S01, S02, S03), North Mahanoy Creek (S06), Waste House Run at St. Nicholas (S08), and Shenandoah Creek (S13, S14). These tributaries were affected by leakage to underground mines that resulted in complete loss of streamflow during low base-flow conditions at sites S01, S02, S03, S06, and S08. Sanders & Thomas, Inc. (1975) identified specific losing stream reaches and surface-water capture zones upstream from these sites. No attempt has been made with this assessment to update this information.

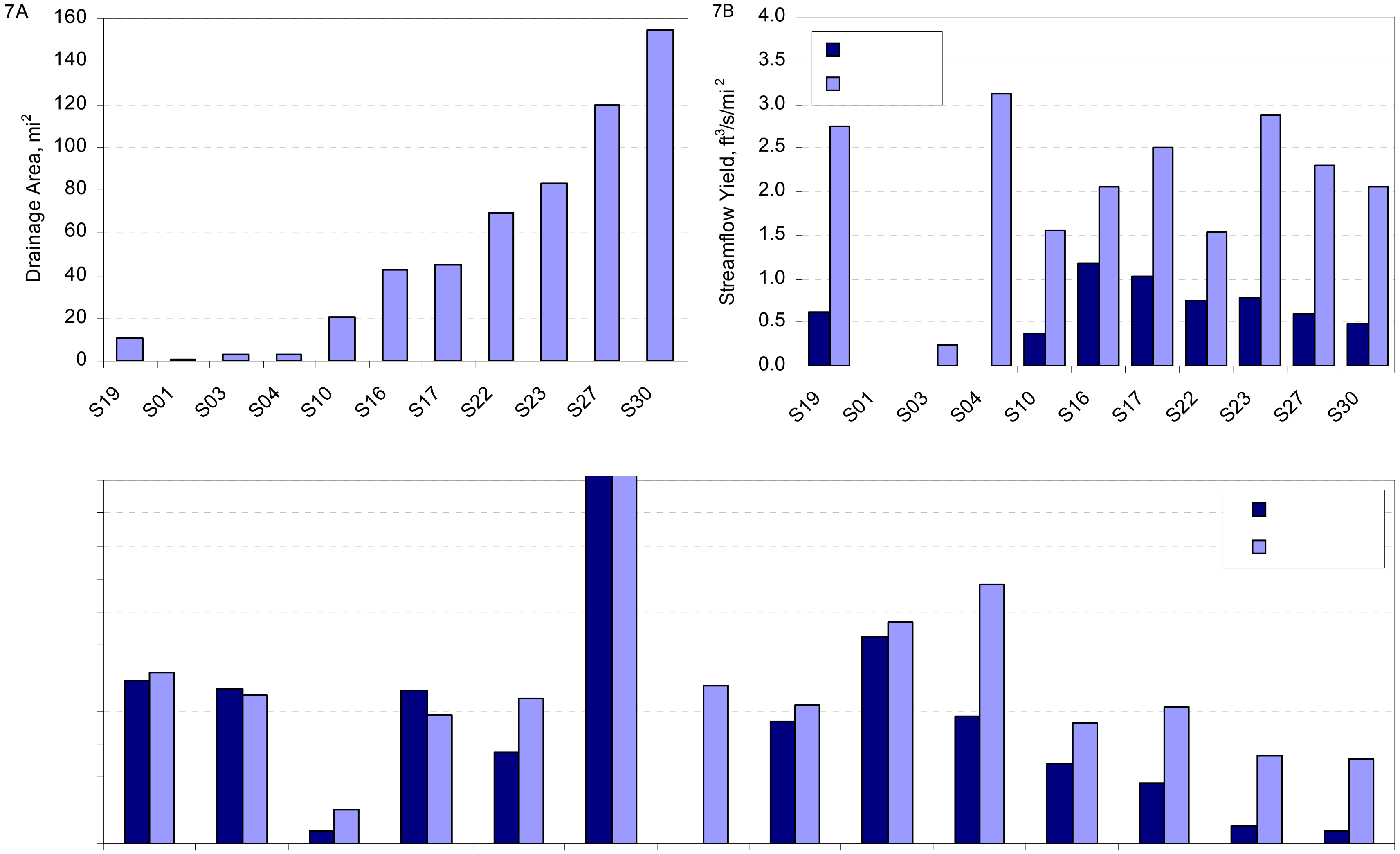

Figure 7. (A) Contributing area in square miles (mi^2), (B) streamflow yields in cubic feet per second per square mile (ft^3/s/mi^2), and (C) dissolved metals loading percentage (%) of abandoned mine drainage (AMD) metals loading at streamwater monitoring sites in the Mahanoy Creek Basin, Pennsylvania, March 2001 and August 2001. Local site numbers shown in figure 1. In 7C, the scale on the ordinate (Y) axis was truncated; values greater than 100 percent indicate the metals loading in streamwater was greater than the sum of metals loading from sampled AMD sources indicated in figure 1 and tables 1 and 2.

Contaminant Concentrations and Loads

To evaluate the potential for transport and attenuation of metals contributed by various AMD sources upstream of stream survey sites, the dissolved metals loads from all known AMD sources were summed and compared with the measured load at the stream site (fig. 7C). Except for Shenandoah Creek near Girardville (S14) and Mahanoy Creek at Ashland (S16) and Gordon (S17), the measured dissolved loads of metals in stream samples downstream from known AMD sources were less than the sum of metals loads from these AMD sources. Lower measured loadings of metals at the stream sites compared to the cumulative load from upstream AMD sources generally results from the precipitation and adsorption of metals downflow from the AMD source(s).

Higher measured loadings of metals at some stream sites (S14, S16, S17) compared to the sum of metals loading from upstream AMD sources (fig. 7C) could indicate that one or more large sources of metals were not sampled upstream from that stream site. For example, access was restricted in the lower part of the Shenandoah Creek subbasin. Thus, some previously reported discharges including the Hammond Mine Connerton Village Boreholes (M09) and Hammond Mine Connerton #1 and #2 discharges (M10) that likely contributed to metals concentrations and streamflow of Shenandoah Creek near Girardville (S14) were not sampled. In contrast, all known large AMD sources were sampled along Mahanoy Creek between Girardville (S10) and Ashland (S16). However, increased metals loading between these two sites was apparent for low and high base-flow conditions indicating the presence of additional large AMD discharges within this reach. The elevation of the stream bottom typically is below the reported ground-water elevations of the underlying mine pools and known discharges (Sanders & Thomas, Inc., 1975). Thus, an upward flow gradient would be expected and the additional unreported AMD sources may discharge within the stream channel. Subsequent to the 2001 surveys, some “new” discharges were identified on the south side of Mahanoy Creek near Ashland (Stephen Ulceski, Ashland Borough, written commun., 2003). Further investigation is needed to determine the locations and characteristics of these unsampled and (or) unidentified discharges.

The attenuation of metals loading as a percentage of the total known AMD loading generally was greater during low base-flow than high base-flow conditions (fig. 7C). The attenuation of dissolved metals within the stream channel during low base-flow conditions can reduce the total metal loading at downstream points. However, resuspension and transport of metals as particles can take place during high base-flow or stormflow conditions. Greater instream loadings at high base-flow conditions indicate that contaminant concentrations were not simply diluted with increased flow, but greater quantities of metals had entered the stream from various sources. Some of the AMD sources could be intermittent (runoff or discontinuous seepage), contributing metals only during high-flow conditions, such as discharges from the Vulcan-Buck Mountain seepage (M02), the Gilberton Mine Pump (M04), and the Bast Mine overflow (M20).

Although the potential toxicity of dissolved aluminum and other metals, such as nickel and zinc, to fish and other aquatic organisms can be reduced by their removal from the water column, the iron, aluminum, and associated metals in AMD can accumulate on or within the streambed forming ochreous (iron-rich) encrustations (Winland and others, 1991; Webster and others, 1998). The accumulation of metals in the streambed will degrade the aquatic habitat and may only temporarily reduce the downstream metal loading. The scour and resuspension of the precipitated metals during high-flow events can result in the non-attainment of water-quality criteria in downstream reaches. This condition has been documented on the basis of stormflow sampling on Swatara Creek (Cravotta and Bilger, 2001) and is indicated to a lesser extent by the high base-flow data of this investigation. Generally, the non-conservative transport of metals from the AMD sources to downstream monitoring sites invalidates simple computations of load reductions required for attainment of TMDLs. Because the metals are not transported conservatively from the AMD source to downstream locations, correction factors would be needed to relate the load reduced at an AMD source to the corresponding load reduction at a downstream location.

Aquatic Ecology

Aquatic ecological surveys that included fish samples were conducted at five stream sites during low base-flow conditions in October 2001 (table 4). The streamwater at these sites had pH greater than 6 during the surveys. All the fish species identified had been previously reported for Pennsylvania streams with pH from 4.6 to 6.5 (table 4). Twenty species of fish were identified in Schwaben Creek near Red Cross, which drains an unmined area of 22.7 mi^2 (58.8 km^2) in the lower part of the Mahanoy Creek Basin. In contrast, 14 species of fish were identified in Mahanoy Creek near its mouth at Kneass, below Schwaben Creek. The diversity and abundance of fish species in Mahanoy Creek decreased progressively upstream from 13 species at Gowen City to only 2 species at Ashland and Girardville. Some of the species in Schwaben Creek and Mahanoy Creek at Kneass or Gowen City are considered intolerant of pollution and acidity, including swallowtail shiner (*Notropis procne*), rosyface shiner (*Notropis rubellus*), river chub (*Nocomis micropogon*), banded darter (*Etheostoma zonale*), and shield darter (*Percina peltata*) (table 4). Cold-water species, including juvenile brown trout (*Salmo trutta*), longnose dace (*Rhinichthys cataractae*), and blacknose dace (*Rhinichthys atratulus*), were captured from Mahanoy Creek at Gowen City (table 4). White sucker (*Catostomus commersoni*), a pollution-tolerant species that feeds on organic matter and organisms within sediments in cold- and warm-water environments, was present at each of the surveyed reaches. The presence of any fish in Mahanoy Creek at Girardville was unexpected because of the poor water quality and iron-encrusted streambed at this

Table 4. Fish species identified and number of individuals counted during ecological survey of Mahanoy Creek and selected tributary streams, Pennsylvania, October 2001[1]

Taxa		Minimum pH in Pa.[2]	Pollution tolerance[3]	Mahanoy Creek				Schwaben Cr nr Red Cross (S28)
ORDER Family *Genus species*	Common name			Girardville (S10)	Ashland (S16)	Gowen City (S23)	Kneass (S30)	
CYPRINIFORMES								
Cyprinidae								
Campostoma anomalum	Stoneroller	6.0	M	0	0	0	4	36
Cyprinella analostana	Spotfin shiner	6.4	M	0	0	1	9	308
Exoglossum maxillingua	Cutlips minnow	6.1	I	0	0	0	0	12
Luxilus cornutus	Common shiner	6.0	M	0	0	0	0	4
Nocomis micropogon	River chub	6.0	I	0	0	1	0	0
Notropis hudsonius	Spottail shiner	6.4	M	0	0	2	232	350
Notropis procne	Swallowtail shiner	6.5	I	0	0	0	4	0
Notropis rubellus	Rosyface shiner	6.0	I	0	0	0	8	21
Pimephales notatus	Bluntnose minnow	5.6	T	0	0	0	13	40
Pimephales promelas	Fathead minnow	6.5	T	1	0	0	0	0
Rhinichthys atratulus	Blacknose dace	5.6	T	0	0	39	0	1
Rhinichthys cataractae	Longnose dace	5.9	I	0	0	3	0	15
Semotilus atromaculatus	Creek chub	5.2	T	0	4	11	0	15
Semolitus corporalis	Fallfish	6.1	M	0	0	2	0	24
Catostomidae								
Catostomus commersoni	White sucker	4.6	T	19	18	51	40	23
Hypentelium nigricans	Northern hog sucker	6.0	I	0	0	0	0	1
SILURIFORMES								
Ictaluridae								
Noturus insignis	Margined madtom	5.9	M	0	0	0	0	5
SALMONIFORMES								
Salmonidae								
Salmo trutta	Brown trout	5.9	M	0	0	2	0	0
PERCIFORMES								
Centrarchidae								
Ambloplites rupestris	Rock bass	6.0	M	0	0	0	4	6
Lepomis cyanellus	Green sunfish	6.4	T	0	0	1	3	0
Micropterus dolomieu	Smallmouth bass	6.0	M	0	0	9	163	60
Percidae								
Etheostoma olmstedi	Tessellated darter	5.9	M	0	0	9	19	16
Etheostoma blennioides	Greenside darter	6.0	M	0	0	3	24	34
Etheostoma zonale	Banded darter	6.0	I	0	0	0	21	31
Percina peltata	Shield darter	6.5	I	0	0	0	1	1
Total number of individuals collected:				**20**	**22**	**134**	**545**	**1,003**
Total number of species identified:				**2**	**2**	**13**	**14**	**20**

1. Fish collected by electrofishing, identified, and released by U.S. Geological Survey on October 10-11, 2001. Counts are indicated below site heading.

2. Minimum pH of occurrence in freshwater in Pennsylvania as reported by Butler and others (1973).

3. Pollution tolerance: I (intolerant), M (moderate), T (tolerant), adapted from Barbour and others (1999).

location. Despite excellent physical habitat offered by deep plunge pools and boulders bordered downstream by riffles, fish were not present at the top of the surveyed reach at Girardville. Near the top of the reach, in the vicinity of the Girard Mine seepage (M11), a strong odor of hydrogen sulfide in the air was detectable because of mixing and aeration of various AMD inflows.

Although poor water quality and iron-rich coatings on the streambed degraded the aquatic habitat in the mining affected reaches of Mahanoy Creek and its tributaries, one or more aquatic macroinvertebrate species (insects, crustaceans, worms) were observed at each of the 17 sites that was surveyed for water chemistry and macroinvertebrate diversity (table 5). Generally, macroinvertebrate diversity and abundance at the mining-affected sites on Mahanoy Creek and tributaries were diminished compared to Little Mahanoy Creek, Mouse Creek, Schwaben Creek, and other unmined tributaries (table 5, figs. 5H and 8). Nevertheless, even if mined, subbasins that were predominantly forested had a greater diversity and a higher water-quality indication on the basis of the Hilsenhoff (1988) family-level biotic index than those with extensive unreclaimed or revegetated mined areas or with extensive agricultural development (fig. 8). For example, the Hilsenhoff biotic index indicated excellent or very good water quality in Mahanoy Creek near Mahanoy City (S03) and at Gowen City (S23) and in Kehly Run near Shenandoah (S11) where second-growth forests surround the immediate area (table 5, figs. 5H and 8). Also, Mouse Creek (S29), where the drainage area is dominated by agricultural land use but also includes substantial forested cover, had very good water quality compared to fair quality of Schwaben Creek (S28) on the basis of the Hilsenhoff biotic index (table 5, fig. 5H and 8). In Mahanoy Creek at Gowen City, 16 benthic macroinvertebrate taxa (family level), including 3 genera each of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) were identified (table 5). However, elsewhere, the most common specimens identified in degraded reaches were Hydropsyche and Chironomidae that are known to tolerate acidic or organic-enriched conditions. For example, within the discharge zone of the Vulcan-Buck Mountain Boreholes (M03) and other AMD affected reaches on Mahanoy Creek (S09, S10, S16), chironomids and other diptera species considered tolerant of pollution were predominant (table 5).

Metals that have accumulated in the sediments of mining-affected streams could become elevated in the tissue of inhabitant fish and other aquatic organisms (Winterbourn and others, 2000; Cravotta and Bilger, 2001). Concentrations of metals in the whole-fish sample of white sucker from Mahanoy Creek near Gowen City were similar to those for equivalent samples collected October 1999 from Shamokin Creek near Shamokin and Swatara Creek at Ravine, Pa. (table 6). Iron, manganese, aluminum, strontium, copper, nickel, and zinc concentrations were elevated compared to other metals in the whole-fish samples from these mined watersheds (table 6). In general, because of metals in gut contents and in organs such as the liver, concentrations of most metals in whole fish will be greater than those in fish prepared for consumption (Campbell and others, 1988; Cravotta and Bilger, 2001). Nevertheless, none of the metals in the whole white sucker samples exceeded U.S. Environmental Protection Agency (1997) screening values for human consumption (table 6). The concentrations of zinc exceeded the national average for whole-fish samples reported by Lowe and others (1985). Copper and selenium were similar to the national averages, and mercury and lead were lower than the national averages (Lowe and others, 1985). The following elements were not detected in the fish samples: antimony (Sb), boron (B), cadmium (Cd), molybdenum (Mo), silver (Ag), and uranium (U) (table 6).

Streambed Chemistry

Ochreous precipitates at the AMD sites in the Mahanoy Creek Basin were composed predominantly of iron and aluminum compounds in the form of goethite (FeOOH), ferrihydrite ($Fe(OH)_3$), schwertmannite ($Fe_8O_8(OH)_6SO_4$), and (or) poorly crystalline or amorphous aluminum-hydroxysulfate minerals (table 7). The iron minerals imparted rusty brownish colors to the sediments. Quartz (SiO_2), muscovite ($KAl_3Si_3O_{10}(OH)_2$), kaolinite ($Al_2Si_2O_5(OH)_4$), and gypsum ($CaSO_4 \cdot 2H_2O$) were locally present at the AMD sites. Downstream from the AMD sources, streambed sediment samples were composed of quartz, muscovite, kaolinite, chlorite ($Mg_5Al_2Si_3O_{10}(OH)_8$), and (or) calcite ($CaCO_3$) along with the iron-bearing minerals (table 7). These streambed sediments were enriched in iron, aluminum, and (or) manganese plus trace metals (table 7). Median concentrations of aluminum, manganese, cadmium, chromium, copper, nickel, lead, and zinc in the fine fraction of streambed sediments (less than 0.180 mm) from Mahanoy Creek and its tributaries exceeded those for the AMD sediments. Only the median concentrations of iron and arsenic in the AMD sediments were greater than those for the streambed samples.

Compared to median concentrations in streambed sediments across the conterminous United States (Rice, 1999), the streambed sediments from Mahanoy Creek and its tributaries had two to three times greater median concentrations of iron, arsenic, cadmium, copper, nickel, lead, and zinc and smaller median concentrations of aluminum and chromium (table 7). Concentrations of iron, manganese, copper, nickel, lead, and (or) zinc in the streambed sediments from Mahanoy Creek commonly exceeded sediment-quality guidelines (SQGs) for protection of sediment-dwelling organisms in freshwater ecosystems (Persaud and others, 1993; MacDonald and others, 2000) (table 7, fig. 9). The SQGs indicate metal concentration levels above which harmful effects to benthic organisms are likely. The “probable effect level” (PEL) guideline applies for iron and manganese (Persaud and others, 1993); the “probable effect concentration” (PEC) guideline applies for arsenic, cadmium, chromium, copper, lead, nickel, and zinc (MacDonald and others, 2000). MacDonald and others (2000) concluded that a geometric mean PEC quotient (concentration/PEC) greater than 0.5 indicated sediment toxicity because of multiple contaminants.

Table 5. Aquatic macroinvertebrate abundances and corresponding Hilsenhoff family-level biotic index at stream water-quality monitoring sites in the Mahanoy Creek Basin, Pa., 2000-2002.[1,2]

[Taxa abundance reported as range: -, not observed; R, rare (1-2); P, present (3-9); C, common (10-24); A, abundant (25-100); VA, very abundant (>100)]

Taxa	Pollution tolerance[3]	Local monitoring site number and date (YYMMDD)																				
ORDER Family *Genus*		M03	S03	S06	S07	S09	**S10**	S11	S12	S13	S14	**S16**	S19	S20	S21	**S23**	S24	S25	S26	**S28**	S29	**S30**
		010821	020509	020404	010820	010822	**010822**	010822	010822	010820	010820	**010820**	010820	010629	010518	**000809**	000809	000809	000809	**000719**	000726	**000808**
EPHEMEROPTERA (mayflies)																						
Baetidae	6	-	-	-	-	-	-	-	-	-	-	-	C	-	-	C	R	-	-	A	P	R
Ephemerellidae	2	-	-	-	-	-	-	-	-	-	-	-	-	-	P	-	-	-	-	-	-	-
Heptageniidae	3	-	-	-	P	-	-	-	-	-	-	-	-	-	R	C	C	-	-	P	A	-
Isonychiidae	3	-	-	-	-	-	-	-	-	-	-	-	R	-	-	-	-	-	-	A	A	R
Leptophlebiidae	4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	P	-	-	-	-	R	-
ODONATA (dragonflies, damselfies)																						
Aeshnidae	3	-	-	-	-	R	-	R	-	-	-	-	-	-	-	-	P	-	-	-	-	-
Cordulegastridae	3	-	-	-	R	-	-	P	-	-	-	-	R	-	-	-	-	-	-	-	-	-
Gomphidae	4	-	-	-	P	-	-	-	-	-	-	-	-	-	-	-	-	-	R	-	-	-
Calopterygidae	5	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PLECOPTERA (stoneflies)																						
Leuctridae	0	-	A	-	R	-	-	C	-	-	-	-	P	-	-	C	C	-	-	-	C	-
Nemouridae	2	-	-	-	-	-	-	-	-	-	-	-	-	-	R	-	-	-	-	-	-	-
Peltoperlidae	2	-	-	-	R	-	-	-	-	-	-	-	-	-	-	C	-	-	-	-	-	R
Perlidae	3	-	-	-	-	-	-	-	-	-	-	-	C	-	-	C	C	-	-	-	R	-
Pteronarcyidae	0	-	-	-	-	-	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	-
COLEOPTERA (aquatic beetles)																						
Elmidae	5	-	-	-	-	P	-	-	-	-	-	-	-	-	-	-	C	-	-	C	A	R
Psephenidae	4	-	-	-	-	-	-	-	-	-	-	-	P	-	-	P	-	-	-	C	C	-
MEGALOPTERA (dobsonflies, alderflies)																						
Crydalidae																						
Corydalus	4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	P	-	R
Nigronia	2	-	-	-	-	R	P	P	-	-	-	-	-	R	P	R	C	-	-	R	P	-
Sialidae	6	-	-	-	-	P	R	-	-	-	R	P	-	-	-	R	-	-	R	R	R	-
TRICHOPTERA (caddisflies)																						
Glossosomatidae	0	-	-	-	-	-	-	-	-	-	-	-	R	-	-	R	-	-	-	-	P	-
Hydropsychidae	5	-	-	-	C	P	R	A	-	-	P	-	A	A	A	A	A	-	R	A	A	A
Limnephilidae	4	-	-	P	R	-	-	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-
Philopotamidae	3	-	-	-	P	-	-	-	-	-	-	-	C	-	-	A	A	-	-	P	A	-
Rhyacophilidae	1	-	-	P	R	-	-	-	-	-	-	-	C	-	-	-	-	-	-	-	-	-
DIPTERA (true flies)																						
Athericidae	2	-	-	-	P	-	-	-	-	-	-	-	-	-	-	-	-	-	-	P	P	P
Ceratopogonidae	6	-	-	-	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	-	-	-
Chironomidae (red)	10	P	-	-	-	P	R	-	-	-	-	R	-	-	-	-	P	-	-	-	-	-
Chironomidae (other)	6	-	-	C	R	C	-	-	-	-	-	-	A	P	VA	P	P	-	-	P	C	R
Simuliidae	6	-	-	C	-	-	-	C	-	-	-	-	-	-	A	-	R	-	-	P	-	-
Tabanidae	6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	R	-	-
Tipulidae	4	-	R	-	P	-	-	R	R	-	R	R	R	-	P	R	R	-	R	-	P	R
Other Diptera	6	R	-	-	-	R	R	R	-	-	-	-	-	R	-	-	-	-	-	-	-	-
NON-INSECT TAXA																						
DECAPODA (crayfish)																						
Cambaridae	6	-	-	-	C	-	-	R	R	-	-	-	R	R	P	P	P	-	-	-	P	-
MOLLUSCA (clams, snails)																						
Physidae	8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	P	-	-	-	-	-	-
AMPHIPODA (scuds)																						
Crangonyctidae	4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	A	-
OLIGOCHAETA (aquatic earthworms)																						
Oligochaeta	10	-	-	R	-	-	-	-	R	-	R	P	P	-	P	-	-	R	-	C	R	-
Tubificidae	10	-	-	-	-	-	-	-	-	VA	-	-	-	-	-	-	-	-	-	-	-	-
PLATYHELMINTHES (flatworms)																						
Turbellaria	9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	A	-	-
Other Worms	9	-	-	-	-	-	-	-	-	-	-	-	-	R	-	-	-	-	-	R	-	-
Hilsenhoff's family-level biotic index:[4]		9.2	0.09	5.24	4.21	6.03	4.38	4.14	6	10	5.75	7.8	4.62	5.14	5.61	3.76	3.75	10	4.75	5.65	3.76	4.68

1. Monitoring sites described in tables 1 and 2. Shaded columns for unmined subbasins. Bold site numbers where fish surveys were completed in October 2001 (table 4).

2. Macroinvertebrates collected and identified by Martin A. Friday of Pennsylvania Department of Environmental Protection using rapid bioassessment protocols (U.S. Environmental Protection Agency 1993; Barbour and others, 1999) during July 2000 to May 2002. Additional macroinvertebrate data were collected for other sites that did not have water-quality data; these data are reported in appendix A and illustrated in figure 5H as the Hilsenhoff (1988) biotic index.

3. Pollution tolerance index values from 0 (intolerant) to 10 (tolerant) and number of organisms for each taxa used to compute the family-level biotic index (Hilsenhoff, 1988): 0.00-3.75 (excellent), 3.76-4.25 (very good), 4.26-5.00 (good), 5.01-5.75 (fair), 5.76-6.50 (fairly poor), 6.51-7.25 (poor), and 7.26-10.00 (very poor). Tolerance values are based on tolerance to organic pollution and are not necessarily indicative of tolerance to acid or metals pollution or sedimentation.

4. Hilsenhoff index computed by assuming the number of organisms was equal to the average for the abundance classification range: - = 0; R = 1.5; P = 6; C = 17; A = 62.5; and VA = 125.

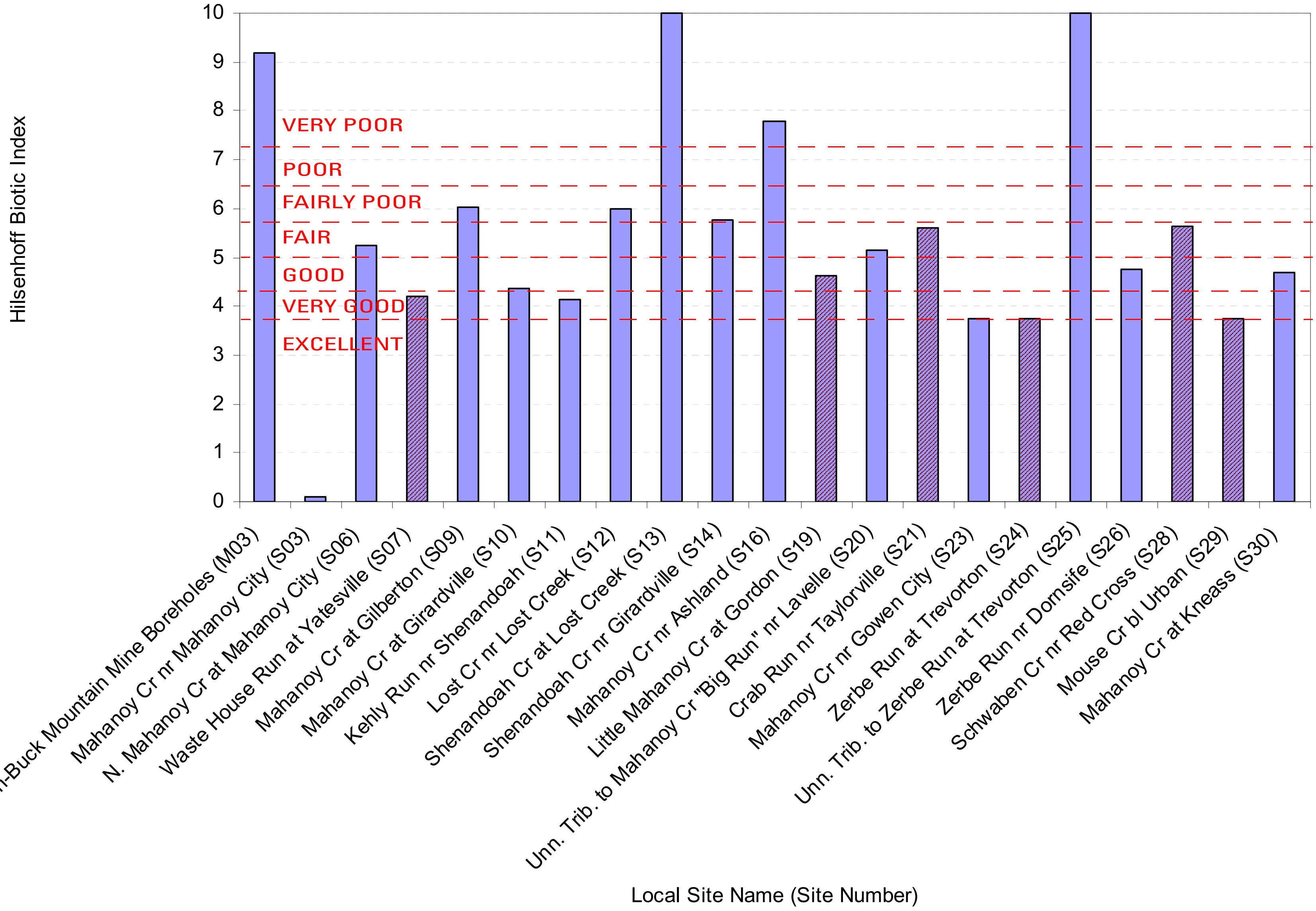

Figure 8. Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity, 2000-2002, at sites sampled for water chemistry in the Mahanoy Creek Basin, Pennsylvania. Diagonal-patterned bars indicate unmined subbasins. Local site numbers shown in figure 1. Macroinvertebrates collected and identified by M. A. Friday (Pennsylvania Department of Environmental Protection, written commun., 2003).

Table 6. Metals concentrations for whole white sucker (*Catostomus commersoni*) collected in October 2001 from Mahanoy Creek near Gowen City, Pa., and October 1999 from Shamokin Creek near Shamokin, Pa., and Swatara Creek at Ravine, Pa.

[Concentrations reported as micrograms per gram (µg/g) dry and wet weight for six-fish composite sample; water and solids content in weight percent (%); <, less than; n.a., not applicable]

Constituent		Mahanoy Creek near Gowen City[1]		Shamokin Creek near Shamokin[a]		Swatara Creek at Ravine[a]		Consumption advisory,[2] dry weight	National Monitoring Program geometric mean concentration,[3] wet weight	
		White sucker, whole		White sucker, whole		White sucker, whole				
		Dry	Wet	Dry	Wet	Dry	Wet		1978-79	1980-81
Aluminum	Al	420	131	44.7	13.9	46.9	11.1	n.a.	n.a.	n.a.
Antimony	Sb	<.10	<.03	<.15	<.05	<.20	<.05	n.a.	n.a.	n.a.
Arsenic	As	.40	.12	<.15	<.05	<.20	<.05	3.0	n.a.	n.a.
Barium	Ba	3.40	1.06	1.73	.54	4.20	.99	n.a.	n.a.	n.a.
Beryllium	Be	.10	.03	<.15	<.05	<.20	<.05	n.a.	n.a.	n.a.
Boron	B	<3.00	<.93	.37	.12	.40	.09	n.a.	n.a.	n.a.
Cadmium	Cd	<.10	<.03	<.15	<.05	<.20	<.05	10.0	n.a.	n.a.
Chromium	Cr	<2.00	<.62	.97	.30	1.90	.45	n.a.	n.a.	n.a.
Cobalt	Co	3.50	1.09	.23	.07	.80	.19	n.a.	n.a.	n.a.
Copper	Cu	4.20	1.31	2.45	.76	3.60	.85	n.a.	0.86	0.68
Iron	Fe	2,500.	778.	187.	58.2	103.	24.3	n.a.	n.a.	n.a.
Lead	Pb	.60	.19	.37	.12	.20	.05	n.a.	.19	.17
Manganese	Mn	230.	71.6	15.8	4.91	70.6	16.7	n.a.	n.a.	n.a.
Mercury	Hg	.036	.011	.068	.021	.100	.024	.6	.11	.11
Molybdenum	Mo	.10	.03	<.15	<.05	<.20	<.05	n.a.	n.a.	n.a.
Nickel	Ni	3.80	1.18	.44	.14	1.20	.28	n.a.	n.a.	n.a.
Selenium	Se	2.80	.87	1.56	.49	2.30	.54	50.	.46	.47
Silver	Ag	<.10	<.03	<.15	<.05	<.20	<.05	n.a.	n.a.	n.a.
Strontium	Sr	65.0	20.2	37.9	11.8	62.8	14.8	n.a.	n.a.	n.a.
Uranium	U	<.10	<.03	<.15	<.05	<.20	<.05	n.a.	n.a.	n.a.
Vanadium	V	.70	.22	<.15	<.05	<.20	<.05	n.a.	n.a.	n.a.
Zinc	Zn	67.0	20.9	39.9	12.4	70.0	16.5	n.a.	.26	.24
Water/Solids		77.0%	23.0%	68.9%	31.1%	76.4%	23.6%	n.a.	n.a.	n.a.

1. Mahanoy Creek near Gowen City, Pa. (S23); site described in table 2. Shamokin Creek near Shamokin, Pa. (USGS station 01554500) described by Cravotta and Kirby (2004a). Swatara Creek at Ravine, Pa. (USGS station 01571820) described by Cravotta and Bilger (2001).

2. Human consumption advisory screening levels from U.S. Environmental Protection Agency (1997). Concentrations are on a dry weight basis.

3. National Monitoring Program data for 1978-81 from Lowe and others (1985). Concentrations are on a wet weight basis.

Table 7. Color, mineralogy, and chemistry of fine sediment[1] and corresponding "sediment-quality guidelines"[2] for streamwater and mine-drainage sample sites, Mahanoy Creek Basin, Pa., August 20-22, 2001.

[XRD, X-ray diffraction; SEL, severe effect level; PEC, probable effect concentration; n.d., no data; <, less than; >, greater than]

Local site identification number[3]	Munsell color of dried sediment		XRD minerals[4]	Iron	Aluminum	Manganese	Arsenic	Cadmium	Chromium	Copper	Nickel	Lead	Zinc	Geometric mean sediment-quality quotient[5]	
	Color name	Hue value chroma		(percent)		(parts per million)								PEC	PEC+SEL
Streamwater (SW, SW-AMD)															
S05	light brownish gray	10 YR 6/2	q,m	4.1	7.6	1,500	14	0.4	62	71	26	120	110	0.38	0.49
S07	dark yellowish brown	10 YR 4/6	q,m	3.3	7.3	2,600	12	1.0	54	71	35	220	250	.53	.66
S09	strong brown	7.5 YR 5/8	g,f	44	2.6	1,600	58	<.1	10	70	12	36	170	.23	.44
S10	dark yellowish brown	10 YR 4/4	f	51	1.5	3,600	38	.5	6	97	44	22	480	.35	.67
S11	light olive brown	2.5 Y 5/3	q,m	2.6	4.8	100	11	<.1	37	20	13	86	38	.17	.19
S12	grayish brown	10 YR 5/2	q,m,ch	2.6	5.9	1,600	8.1	.6	55	40	25	57	230	.33	.42
S13	strong brown	7.5 YR 5/8	q,m	4.6	6.5	740	15	2.3	110	390	80	220	960	1.18	1.11
S14	strong brown	7.5 YR 5/8	q,m,k,g	16	6.2	4,600	74	1.0	65	120	100	84	460	.84	1.19
S15	strong brown	7.5 YR 5/8	f	31	3.1	170	13	<.1	39	85	14	34	94	.22	.31
S16	strong brown	7.5 YR 5/8	f,m,q	49	.80	16,000	1	2.0	2	33	320	16	1,300	.27	.65
S17	strong brown	7.5 YR 5/8	q,m,k,g	26	4.6	1,200	33	1.3	72	120	83	82	720	.81	1.06
S19	dark yellowish brown	10 YR 4/4	q,m,ch	3.2	6.5	2,200	10	2.2	70	62	52	120	340	.59	.70
S20	dark olive brown	2.5 Y 3/3	q,m	15	3.9	18,000	48	.9	45	63	200	78	320	.70	1.19
S21	dark yellowish brown	10 YR 3/4	q,m	3	6.2	2,200	10	.6	58	32	38	37	160	.32	.43
S22	strong brown	7.5 YR 5/8	q,m	8.7	3.8	2,600	18	.6	53	51	76	54	330	.47	.67
S23	dark olive brown	2.5 Y 3/3	q,m	16	5.2	8,000	24	1.6	57	96	180	62	860	.82	1.25
S24	dark brown	7.5 YR 3/4	q,m,ch,c	4.1	7.0	2,200	15	1.2	85	75	57	290	440	.73	.85
S25	strong brown	7.5 YR 5/8	q,m,k	5.8	6.5	1,800	21	.9	75	82	120	84	380	.66	.80
S26	strong brown	7.5 YR 4/6	q,m,k	5.8	4.6	180	13	.1	48	56	28	42	120	.25	.29
S27	yellowish brown	10 YR 5/6	q,m	11	5.0	12,000	18	1.7	49	95	300	55	960	.84	1.27
S28	light yelllowish brown	2.5 Y 6/3	q,m	2.6	4.7	700	9.3	.2	41	20	26	24	88	.20	.25
S29	dark gray	5 Y 4/1	q,m	1.9	3.1	540	6.4	.3	36	10	22	23	90	.17	.21
S30	strong brown	7.5 YR 5/8	q,m	13	5.1	9,400	19	1.9	45	100	220	51	980	.81	1.23
S31	light olive brown	2.5 Y 5/4	q,m	7.7	4.4	5,000	14	1.1	42	63	200	39	590	.59	.84
Abandoned Mine Drainage (GW, GW-AMD)															
M03	dark yellowish brown	10 YR 4/4	g,q,m	32	3.1	110	26	0.2	45	88	12	120	95	0.32	0.40
M07	dark yellowish brown	10 YR 3/6	g,q	55	.35	720	300	.2	4	4	44	7.9	310	.20	.37
M11	strong brown	7.5 YR 5/8	g,f,q	45	.34	430	20	.2	4	20	54	21	390	.21	.35
M11	yellowish red	5 YR 5/6	f	49	<.01	810	180	<.1	<1	<3	19	7.1	120	.10	.22
M11	strong brown	7.5 YR 5/8	f	51	<.01	2,300	180	<.1	<1	<3	19	7.2	130	.10	.25
M11	light gray	2.5 Y 7/2	f	50	<.01	6,300	4	.2	<1	3	47	1.6	420	.07	.21
M11	yellowish brown	10 YR 5/8	g,s	52	<.01	440	120	<.1	<1	5	38	7.3	140	.12	.23
M12	yellowish brown	10YR 5/8	g,q	49	.46	17,000	120	.2	7	9	110	9.4	160	.23	.57
M13	strong brown	7.5 YR 5/8	g	53	.18	2,500	180	<.1	2	5	72	2.3	10	.09	.22
M14	strong brown	7.5 YR 5/8	g,q	52	.77	1,500	110	<.1	9	20	68	15	30	.19	.37
M17	dark reddish brown	5 YR 3/4	g	42	1.50	170	45	2.2	2	39	18	5.4	480	.24	.35
M17	yellowish brown	10 YR 5/6	q,f	41	.78	180	21	.8	2	20	9.5	1.9	220	.12	.20
M18	yellowish brown	10 YR 5/6	g,q	45	2.8	370	96	<.1	20	140	9.7	20	170	.27	.42
M19	strong brown	7.5 YR 4/6	g,q,m	35	1.4	83	12	<.1	18	71	7.3	39	63	.17	.24
M20	grayish brown	2.5 YR 5/2	f	46	2.4	470	65	1.0	4	30	50	6.6	420	.28	.45
M21	yellowish red	5 YR 5/6	f,g	49	2.7	9,500	86	.8	2	56	130	10	590	.36	.76
M22	dark olive brown	2.5 Y 5/4	g	46	3.0	390	64	<.1	1	41	18	20	120	.15	.26
M24	"white slime"	n.d.	q,g,a	4.1	>20.	120	6.8	.1	18	98	27	21	40	.17	.19
M24	dark olive brown	2.5 Y 3/3	g,q	55	.46	1,500	50	.1	8	6	36	4.7	160	.14	.29
M25	light olive brown	2.5 YR 5/3	g,f	36	7.1	95	31	.2	7	120	9.1	39	70	.21	.29
M26	light olive brown	2.5 Y 5/3	g,f	36	7.2	110	30	.2	7	120	8.6	39	68	.20	.29
M26	dark yellowish brown	10 YR 4/4	q,m,k,g	13	4.8	620	77	.2	48	49	44	29	82	.34	.46
M27	dark olive brown	2.5 Y 3/3	g,f	56	.03	3,800	33	<.1	<1	<3	82	<.2	31	.05	.15
M28	reddish yellow	7.5 YR 6/8	g,s	47	.17	23	2	<.1	5	20	1.8	71	47	.08	.12
M29	yellowish brown	10 YR 5/8	f,q	47	3.0	13,000	7	1.5	5	92	360	10	1,300	.43	.90
M30	brownish yellow	10 YR 6/8	q,s	30	2.6	4,100	11	.6	18	190	71	330	210	.55	.90
M31	strong brown	7.5 YR 5/8	g	54	1.1	390	28	<.1	12	140	7.7	37	330	.25	.40
M32	strong brown	7.5 YR 5/8	f,g	44	2.8	1,500	38	<.1	6	53	17	50	170	.22	.41
M34	strong brown	7.5 YR 5/8	g,q	46	.27	17	19	<.1	16	400	2.7	6.4	83	.16	.19
SEL, PEC:				4	n.d.	1,100	33.0	5.0	111	149	48.6	128	459	n.d.	n.d.
NAWQA:[6]				3.5	6.4	n.d.	6.3	.4	64	27	27	27	110	n.d.	n.d.

[1] Metal concentrations on dry weight basis in percent or parts per million (1% = 10,000 ppm) for <100 mesh (<0.182-mm) size fraction.

[2] The probable effect concentration (PEC) or severe effect level (SEL) is a sediment-quality guideline above which harmful effects to benthic organisms are likely to be observed (Persaud and others, 1993; MacDonald and others, 2000). SEL for iron and manganese (Persaud and others, 1993); PEC for arsenic, cadmium, chromium, copper, lead, nickel, and zinc (MacDonald and others, 2000).

[3] Site descriptions in tables 1 and 2. Shaded rows indicate sites in unmined subbasins. Site locations shown in figure 1. Complete data for sampled sites in appendix A.

[4] Mineralogy identified by XRD: g=goethite, f=ferrihydrite, s=schwertmannite, a=amorphous aluminum hydroxysulfate, q=quartz, m=muscovite, k=kaolinite, ch=chlorite, c=calcite, gy=gypsum. Low peak-to-background ratios for many of the XRD patterns indicate amorphous material was present in addition to the minerals identified (J. M. Hammarstrom, U.S. Geological Survey, written commun., 2004). Small quantities of a mineral (less than 5 percent) or poorly crystalline or amorphous materials may not be detected by XRD.

[5] Geometric mean of the ratios of the observed concentration and the corresponding PEC or SEL for metals in streambed sediments (after MacDonald and others, 2000).

[6] Median concentration for streambed sediment across the conterminous United States from the National Water-Quality Assessment (NAWQA) Program (Rice, 1999).

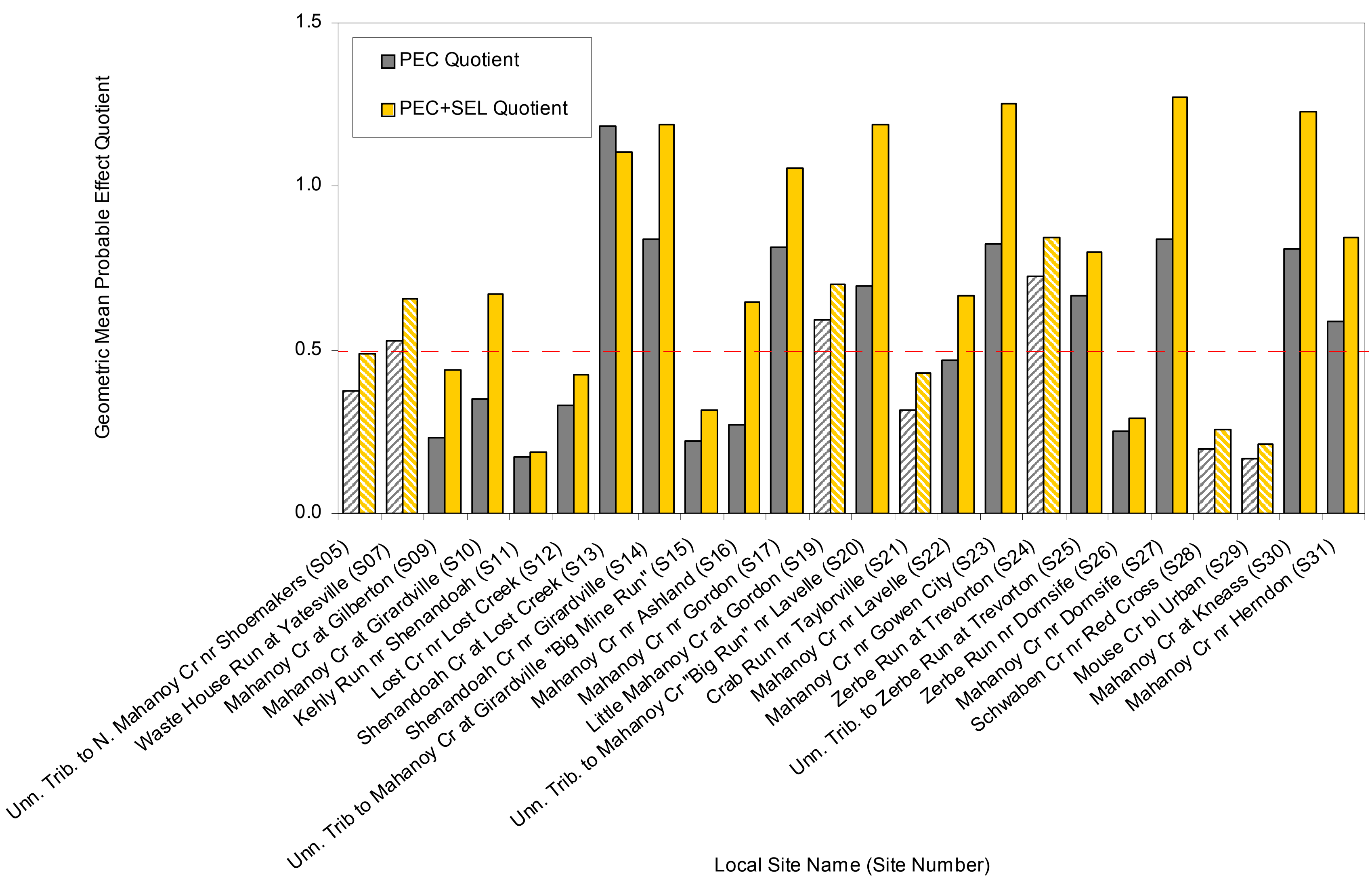

Figure 9. Geometric mean of the ratios of observed and probable effect concentrations (PECs) or PECs plus severe effect levels (SELs) for metals in streambed sediments in the Mahanoy Creek Basin, Pennsylvania, August 2001. Local site numbers shown in figure 1. Diagonal-patterned bars indicate unmined subbasins. The SEL is the guideline for iron and manganese (Persaud and others, 1993) and the PEC is guideline for arsenic, cadmium, chromium, copper, lead, nickel, and zinc (MacDonald and others, 2000). A geometric mean probable effect quotient greater than 0.5 indicates harmful effects to benthic organisms are likely.

Streambed sediments sampled in August 2001 at about half the mining-affected sites (S13, S14, S17, S20, S22, S23, S25, S27, S30, and S31) and half the nonmining sites (S07, S19, and S24) had geometric mean PEC quotients greater than 0.5 reflecting elevated concentrations of copper, nickel, lead, and (or) zinc (fig. 9). In contrast, only 1 of 29 AMD sediment samples (3 percent) had a geometric mean PEC quotient greater than 0.5 (table 7).

The most degraded streambed sediments were from Shenandoah Creek at Lost Creek (S13) and near Girardville (S14). At S13 and S14, metals concentrations in the fine fraction exceeded SQGs for iron (4 wt%), manganese (1,100 mg/g), copper (149 mg/g), nickel (48.6 mg/g), lead (128 mg/g), and zinc (459 mg/g) (table 7). Concentrations of arsenic, cadmium, and chromium were below the PEC or PEL. The geometric mean PEC quotients considering arsenic, cadmium, chromium, copper, lead, nickel, and zinc were 1.18 and 0.84 at S13 and S14, respectively, and comparable PEC+PEL quotients, which included Fe and Mn, were 1.11 and 1.19. Generally, the PEC+PEL quotients were greater than the PEC quotient (table 7) because of elevated concentrations of iron and manganese in the sediments at most streamwater and AMD sites.

Concentrations of iron and aluminum in the AMD and streambed sediments were inversely correlated, and manganese was not correlated with iron or aluminum, indicating one of these metals was predominant in the precipitate at each site. Spearman's rank correlation coefficients were significant between iron and arsenic; among aluminum, cadmium, chromium, copper, and lead; and among manganese, cadmium, cobalt, nickel, strontium, and zinc. Dissolved trace metals have been reported to adsorb to hydrous iron, aluminum, and manganese oxides, and manganese oxides generally are more effective sorbents than iron or aluminum oxides at near-neutral and lower pH (McKenzie, 1980; Coston and others, 1995; Webster and others, 1998; Tonkin and others, 2004). Compared to other trace metals, chromium, copper, and lead tend to adsorb to aluminosilicates (clay) and iron oxides at relatively low pH (4 to 6) (McKenzie, 1980; Webster and others, 1998). In contrast, near-neutral pH values favor adsorption of manganese, cobalt, nickel, and zinc on iron oxides (McKenzie, 1980). Ultimately, the adsorbed manganese can oxidize forming manganese oxides (Cravotta and Trahan, 1999). The general correlation among manganese, cobalt, nickel, and zinc in mine-drainage sediments was previously reported by Cravotta and Trahan (1999) for precipitate that formed within an "oxic limestone drain" and by Cravotta and Bilger (2001) for streambed samples from Swatara Creek in the Southern Anthracite Field of Pennsylvania. Correlations between iron and arsenic in AMD precipitate were previously noted by Cravotta and others (2001).

In summary, despite AMD loading along most of its length, the quality of water in Mahanoy Creek improved downstream of Ashland because of attenuation processes including dilution, precipitation of hydrous iron, aluminum, and manganese oxides, and adsorption of trace metals. The accumulated metals in the streambed of Mahanoy Creek and its mining-affected tributaries degrade the aquatic habitat and serve as a long-term source of metals that can be resuspended during storms and associated high-flow conditions and possibly redissolved if subjected to acidic or anoxic conditions. The lack of taxa richness and trophic imbalance in mining-degraded reaches within the Mahanoy Creek Basin are consistent with the identified toxic effect levels for iron, manganese, nickel, copper, lead, and zinc in streambed sediments and imply that contaminant loadings from AMD sources are stressful to aquatic macroinvertebrates and, consequently, can be limiting to insectivorous (insect-eating) and piscivorous (fish-eating) species of fish and other higher trophic level animals. Elevated concentrations of iron, manganese, aluminum, strontium, copper, nickel, and zinc in whole white sucker sampled from Mahanoy Creek near Gowen City indicate potential for sediment-derived metals to accumulate in aquatic organisms.

Characterization and Remediation of Abandoned Mine Drainage

Numerous AMD sources have been identified as sources of acidity, metals, and other contaminants in Mahanoy Creek and its tributaries. However, the effects on streamwater quality can vary depending on the characteristics of the AMD sources and the receiving stream. Hence, the primary goals of this study were to assess and rank the effects of individual AMD sources and to identify possible remedial alternatives, including passive-treatment options, that could be applied in the basin.

Characteristics of Abandoned Mine Drainage Sources

The water-quality data for high base-flow samples collected in March 2001 and low base-flow samples collected in August 2001 are summarized below for the largest AMD sources on the basis of flow volume, in approximate east-to-west (or downstream) order. The large discharges are identified by local site number indicated in table 1 and distinguished by larger symbols in figure 10 on the basis of their flow rate, acidity (net alkalinity), and metal loadings. Most of the large AMD sources are along the valley bottom near perennial streams. Many AMD sources contribute substantially to base flow of the receiving stream. The tributary subbasin that receives the AMD is identified in table 2. On the basis of previous reports by Sanders & Thomas, Inc. (1975) and Reed and others (1987), the approximate recharge area for each of the AMD sources has been delineated along with other mine features as part of the GIS developed for the project. Possible treatment alternatives on the basis of the AMD water quality (fig. 2) and the proximity of the AMD source to nearby streams, roads, and other land-use features have been noted below for consideration by resource managers and land owners that may be involved in decisions to implement remediation. No attempt has been made in this study to evaluate the feasibility for remediation or treatment of the AMD sources.

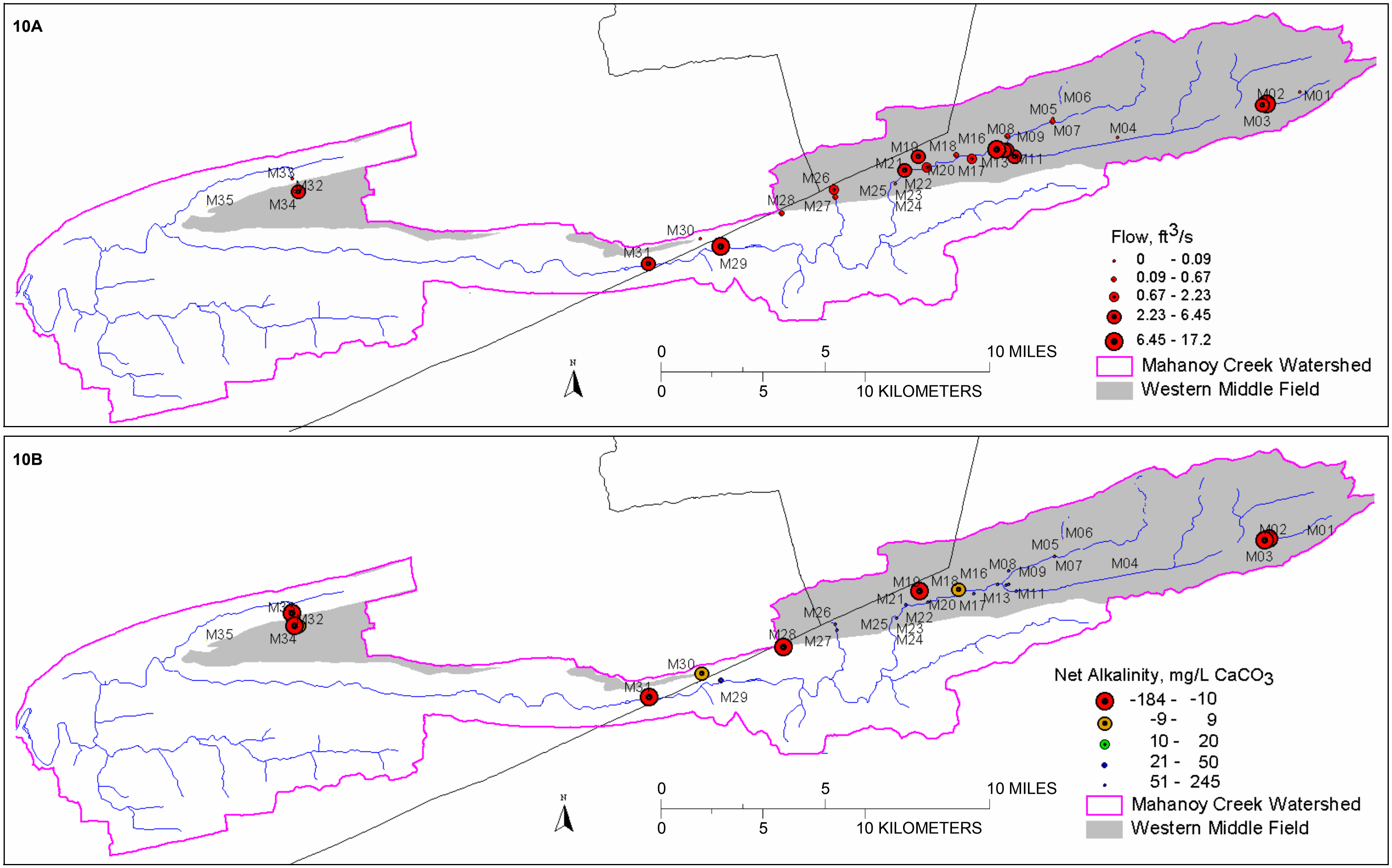

Figure 10. Mahanoy Creek Basin, Pennsylvania, showing abandoned mine drainage locations and corresponding (A) flow rate, (B) net alkalinity concentration, and loading of dissolved metals: (C) iron, (D) manganese, (E) aluminum, and (F) iron+manganese+aluminum, March 2001. Flow rate in cubic feet per second (ft^3/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L $CaCO_3$); loading in megagrams per year (Mg/yr).

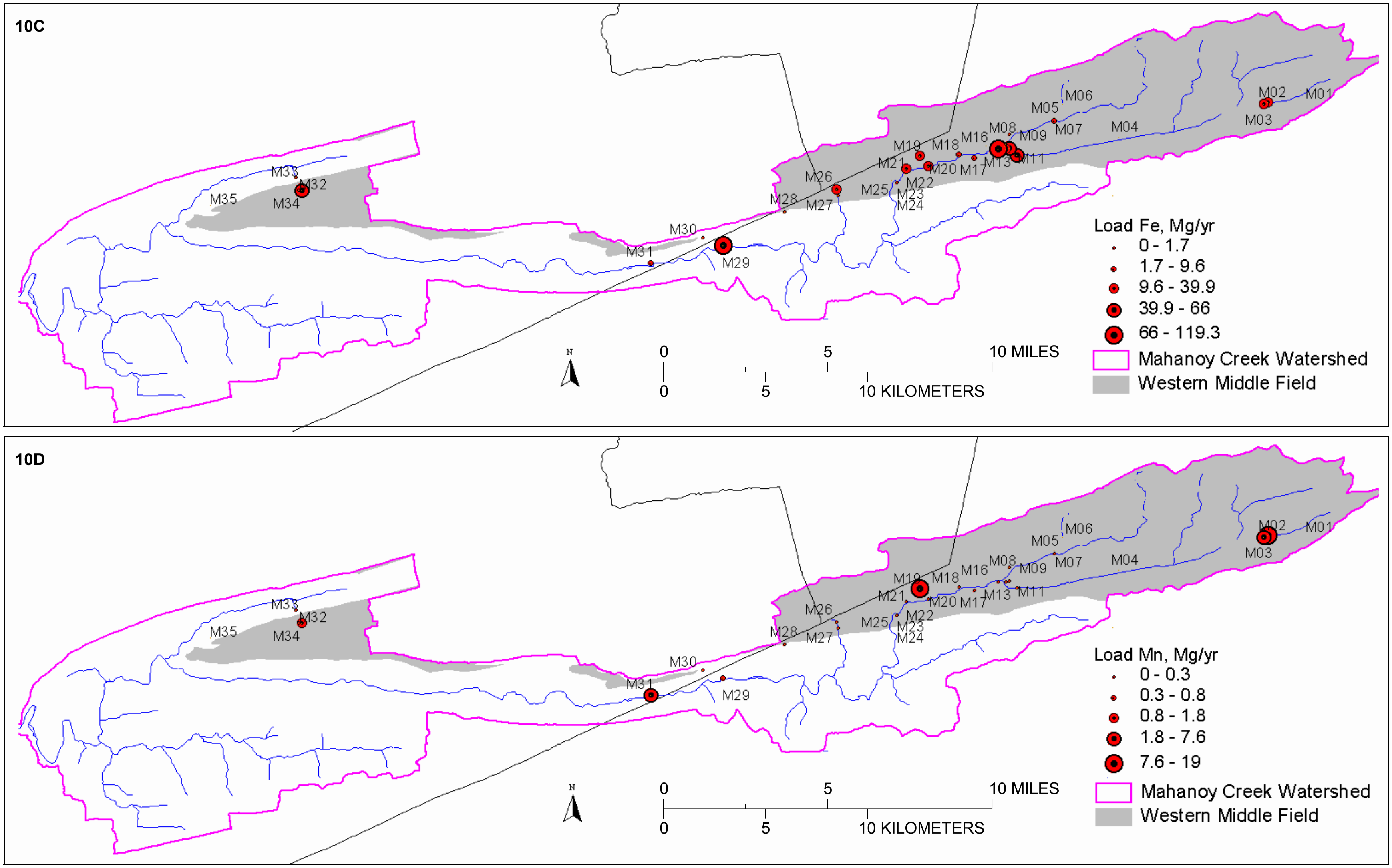

Figure 10. Mahanoy Creek Basin, Pennsylvania, showing abandoned mine drainage locations and corresponding (A) flow rate, (B) net alkalinity concentration, and loading of dissolved metals: (C) iron, (D) manganese, (E) aluminum, and (F) iron+manganese+aluminum, March 2001. Flow rate in cubic feet per second (ft^3/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L $CaCO_3$); loading in megagrams per year (Mg/yr).—Continued

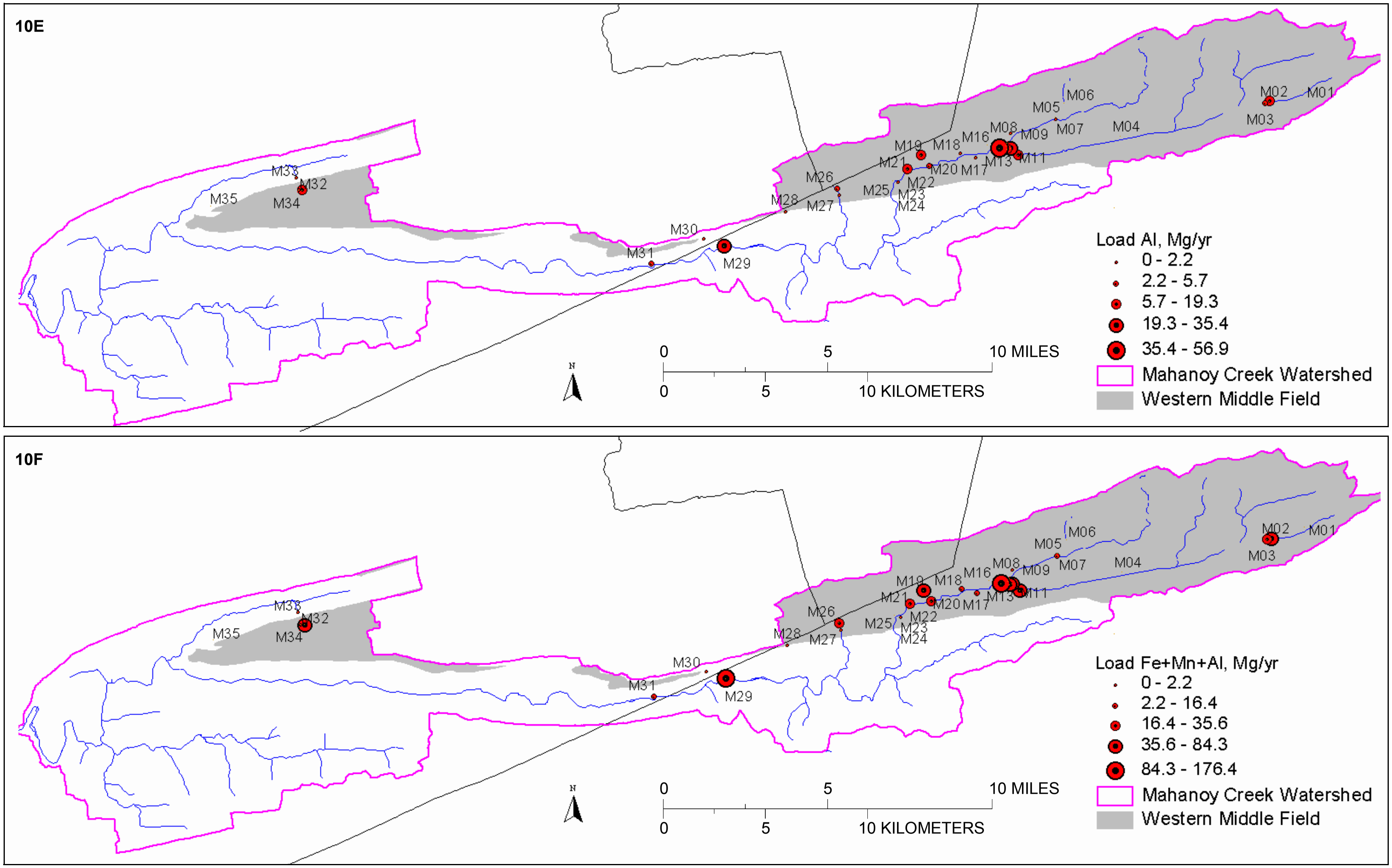

Figure 10. Mahanoy Creek Basin, Pennsylvania, showing abandoned mine drainage locations and corresponding (A) flow rate, (B) net alkalinity concentration, and loading of dissolved metals: (C) iron, (D) manganese, (E) aluminum, and (F) iron+manganese+aluminum, March 2001. Flow rate in cubic feet per second (ft^3/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L $CaCO_3$); loading in megagrams per year (Mg/yr).—Continued

The Vulcan-Buck Mountain Morris Tunnel (M01), the Vulcan-Buck Mountain seepage (M02), and the Vulcan-Buck Mountain Boreholes (M03) all discharge AMD from the Vulcan-Buck Mountain Mine to the headwaters of Mahanoy Creek east of Mahanoy City (fig. 10). Flow is intermittent at M01 and M02 but continuous at M03. During March and August 2001, no water was flowing from site M01, reflecting drought conditions during the study. Water-quality data at M01 previously were reported by Sanders & Thomas, Inc. (1975, site 1) and Reed and others (1987, site 61) indicating it had pH of 3.3, low concentrations of iron and manganese (less than 2 mg/L), and a high concentration of aluminum (greater than or equal to 20 mg/L). In March 2001, discharge at M02 was 8.79 ft^3/s (14,900 L/min) with pH of 4.7, net alkalinity of -21 mg/L, and concentrations of dissolved iron of 3.4 mg/L, aluminum of 2.1 mg/L, manganese of 1.2 mg/L, and oxygen of 0.2 mg/L (table 3, fig. 10). The combined load of dissolved iron, aluminum, and manganese at M02 was 52.6 Mg/yr in March 2001; however, this site was dry in August 2001. In March and August 2001, discharge from the Vulcan-Buck Mountain Boreholes at M03 was 5.28 and 2.64 ft^3/s (8,970 and 4,490 L/min), respectively (table 3). The quality of the AMD was relatively constant with pH of 4.9 and 4.8, net alkalinity of -18 mg/L, and concentrations of dissolved iron of 3.6 and 6.3 mg/L, aluminum of 1.6 and 1.1 mg/L, manganese of 1.2 mg/L, and oxygen of 0.1 and 0.2 mg/L. In March and August 2001, the combined load of dissolved iron, aluminum, and manganese was 30.2 and 20.3 Mg/yr, respectively (table 3, fig. 10). These data indicate that water discharged from the Vulcan-Buck Mountain Mine is net acidic and contributes substantial quantities of metals to Mahanoy Creek. However, treatment would be difficult because the flow varies substantially and area is limited for construction of a treatment system(s). The AMD abatement plan of Sanders & Thomas, Inc. (1975) proposes stream restoration and backfilling of abandoned pits to minimize recharge to the mine.

The Gilberton Mine Pump (M04), which discharges AMD intermittently from the deepest mine in the upper Mahanoy Creek Basin, was not operating during March and August 2001. Reed and others (1987) reported that the Gilberton Mine Pump "operates 40 percent of the time to control water levels and prevent flooding of basements in Gilberton." According to Edward Wytovich (Eastern Pennsylvania Coalition for Abandoned Mine Reclamation, oral commun., 2002), this AMD source was not flowing during any month in 2001, reflecting drought conditions during the study. When the Gilberton Pump is in operation, the discharge drastically affects the streamflow and water quality of Mahanoy Creek from Gilberton downstream to its confluence with Shenandoah Creek. Previously published data indicate the discharge at M04 can be large (23 ft^3/s or 39,100 L/min) and acidic (net alkalinity less than -100 mg/L) because of high concentrations of iron (greater than or equal to 50 mg/L) (Sanders & Thomas, Inc., 1975; Reed and others, 1987). Because the flow rate is extremely variable and the quality is poor, passive treatment of the Gilberton Pump discharge is not a suitable alternative. According to Gary Greenfield (Pennsylvania Department of Environmental Protection, oral commun., 2002), the PaDEP has developed preliminary plans to install a new borehole to the Gilberton Mine at a lower surface elevation than the pump discharge that ideally would maintain the required mine pool elevation and allow AMD to flow continuously by gravity through a constructed wetland.

Downstream from the Gilberton Mine Pump, the Girard Mine seepage (M11) overflows to Mahanoy Creek from the Girard Mine pool near the eastern limits of Girardville (fig. 1). The seepage is predominantly from a collapsed mine opening at M11 along the southern bank of Mahanoy Creek, but various smaller seeps also discharge in the vicinity (Sanders & Thomas, Inc., 1975). In March and August 2001, discharge at M11 was 4.10 and 2.73 ft^3/s (6,970 and 4,640 L/min) with pH of 6.1 and 6.0, net alkalinity of 51 and 15 mg/L, and concentrations of dissolved iron of 18 and 24 mg/L, aluminum of less than 0.01 mg/L, manganese of 3.8 and 4.4 mg/L, and oxygen of 2.2 and 0.3 mg/L, respectively (table 3, fig. 10). In March and August 2001, the combined load of dissolved iron, aluminum, and manganese from M11 was 79.9 and 69.3 Mg/yr, respectively. Although the net-alkaline quality of the Girard Mine seepage is suitable for treatment with aerobic wetlands (fig. 2), passive treatment of the large flow at M11 would not be suitable because of its proximity to Mahanoy Creek and State Route 54 and the large size required for a passive system. Active-treatment options, such as heterogeneous catalysis of iron oxidation by ferric hydroxide, have been proposed (Dietz and Dempsey, 2002).

The Weston Mine surfaces area seepage (M05), Weston Mine Lost Creek-Buck Mountain discharge (M06), and the Weston Mine Lost Creek Borehole (M07) discharge AMD from the upper part of the 11.3-mi^2 (29.3-km^2) Shenandoah complex between Mahanoy City and Girardville (Reed and others, 1987). Variable flow rates have been measured in the Weston Mine discharges; flow is intermittent at M05 and M06 but continuous at M07. During March 2001, AMD from M05 was flowing at only 0.01 ft^3/s (17 L/min) with pH of 6.4 and dissolved oxygen of 9.7 mg/L; the site was dry during August 2001 (table 3). Nevertheless, previous data for M05 indicate a substantial flow rate of 3.7 ft^3/s (6,290 L/min) with pH of 6.1 and iron concentration of 20 mg/L (Reed and others, 1987). The Weston Mine Lost Creek-Buck Mountain discharge at M06 could not be found and was not sampled during the current study; this discharge probably was dry. Previous data indicate that this discharge had a pH of 4 but relatively low concentrations of dissolved metals (less than 2 mg/L) (Sanders and Thomas, Inc., 1975). In March and August 2001, discharge at M07 was 0.38 and 0.03 ft^3/s (646 and 51 L/min), respectively, with pH of 6.1, net alkalinity of 57 and 11 mg/L, and concentrations of dissolved iron of 14 and 33 mg/L, aluminum of less than 0.01 mg/L, manganese of 6.4 and 8.9 mg/L, and oxygen of 0.3 and 0.4 mg/L (table 3, fig. 10). In March and August 2001, the combined load of dissolved iron, aluminum, and manganese from the Weston Mine discharges was 6.9 and 1.2 Mg/yr, respectively (table 3).

The Hammond Mine seepage (M08), Hammond Mine Connerton Village Boreholes (M09), and the Hammond Mine Connerton #1 and #2 discharges (M10) discharge AMD from the middle part of the Shenandoah complex between Mahanoy City and Girardville (Sanders & Thomas, Inc., 1975; Reed and others, 1987). The Hammond Mine discharges have variable flow rates; flow is intermittent at M08 but probably continuous at M09 and M10, which are at lower elevations than M08 along Shenandoah Creek. Discharge from the Hammond Mine enters a large wetland and accounts for tripling of the flow rate of Shenandoah Creek between sites S13 and S14 (fig. 1, table 3). In March 2001, discharge at the Hammond Mine seepage (M08) was 0.27 ft^3/s (459 L/min) with pH of 6.6, net alkalinity of 164 mg/L, and concentrations of dissolved iron of 5.2 mg/L, aluminum of less than 0.01 mg/L, manganese of 3.5 mg/L, and oxygen of 0.5 mg/L (table 3, fig. 10). The combined load of dissolved iron, aluminum, and manganese at M08 was 2.1 Mg/yr. Because access was restricted along Shenandoah Creek in the area of the wetland, AMD sources at M09 and M10 could not be verified or sampled for this study; these may be one and the same AMD source, but the mapped locations and site descriptions in Sanders & Thomas, Inc. (1975) and Reed and others (1987) are not identical. Although current data on the flow rates and quality of the Connerton discharges are lacking, passive treatment of these and the Hammond Mine seepage is taking place in the wetland. The AMD abatement plan of Sanders & Thomas, Inc. (1975) that proposes stream restoration and backfilling of abandoned pits to minimize recharge to the Shenandoah complex could reduce flow rates from the Hammond Mine and, consequently, increase detention time and treatment effectiveness within the wetland.

The Packer #5 Mine discharges are the largest sources of AMD from the 11.3-mi^2 (29.3 km^2) Shenandoah complex between Mahanoy City and Girardville (Reed and others, 1987). The Packer #5 Mine discharge sampling sites (M12, M13, and M14) are within the municipal boundaries of Girardville (fig. 1). The discharges from the Packer #5 Mine Borehole (M12) and Breach (M13), which are approximately 0.1 mi (0.16 km) downstream from M12, are collected into a 0.50 mi (0.80 km) long ditch that is diverted through a culvert under Mahanoy Creek. The combined flow from the Packer #5 Mine Borehole and Breach within the ditch (M14) enters Mahanoy Creek approximately 0.25 mi (0.40 km) below the confluence of Mahanoy Creek and Shenandoah Creek. Although the flow at M14 consists entirely of drainage from M12 and M13, data for M14 were shown with streamwater samples in tables 2 and 3 and figure 3 because of the physical setting of this site. The chemistry of samples from M12, M13, and M14 is similar (table 3). In March and August 2001, discharge at M14 was 10.27 and 9.42 ft^3/s (17,400 and 16,000 L/min) with pH of 6.4, net alkalinity of 95 and 76 mg/L, and concentrations of dissolved iron of 13 and 21 mg/L, aluminum of 0.02 and 0.01 mg/L, manganese of 6.2 and 9.2 mg/L, and oxygen of 3.3 and 2.0 mg/L, respectively (table 3, fig. 10). The combined load of dissolved iron, aluminum, and manganese at M14 was 176 and 254 Mg/yr in March and August 2001, respectively.

The metal loading rate from the combined Packer #5 Mine discharges (M14 or M12 plus M13) is greater than that from any other AMD source in the Mahanoy Creek Basin (table 3). The net-alkaline quality of the AMD from the Packer #5 Mine is consistent with data reported by Sanders and Thomas, Inc. (1975) and Reed and others (1987) and indicates aeration and settling basins may be appropriate for treatment (fig. 2). However, larger flow rates ranging from 38 to 49 ft^3/s (64,600 and 83,300 L/min) were documented by Reed and others (1987). Because of the large flow and metals loading rates, the construction of aerobic wetlands will require a large area (more than 7 acres). Reed and others (1987) indicated that infiltration to the Shenandoah complex is 25 percent higher than other areas in the Mahanoy Creek Basin. According to Sanders & Thomas, Inc. (1975), the streamflow from Waste House Run, Lost Creek, and other unnamed tributaries enters open pits and (or) infiltrates, accounting for approximately 12 percent of the AMD discharged from the Shenandoah complex. The AMD abatement plan of Sanders & Thomas, Inc. (1975) proposes stream restoration and backfilling of abandoned pits to minimize recharge to the Shenandoah complex and, consequently, to reduce flow rates from the Packer #5 Mine.

The Preston Mine discharges from three locations (M15, M16, and M17) to Mahanoy Creek within the municipal boundaries of Girardville (fig. 1). Neither the Preston Mine #3 water level drift (M15) nor the Preston Mine #2 overflow (M16) was sampled during this study. Site M16 is within the wetland on Shenandoah Creek and could not be accessed. Although the published locations for M15 and M16 differed between Sanders & Thomas, Inc. (1975) and Reed and others (1987), these AMD sources may be one and the same based on their descriptions and water quality. The published coordinates for M17 in Reed and others (1987) are erroneous; however, the correct location of the Preston Mine #3 Tunnel overflow (M17) was shown on hand-marked maps provided by L.A. Reed (U.S. Geological Survey, written commun., 2001), and these coordinates are shown in table 1. In March and August 2001, discharge at M17 was 2.23 and 0.67 ft^3/s (3,790 and 1,140 L/min) with relatively constant pH of 6.3 and 6.2, net alkalinity of 59 and 72 mg/L, and concentrations of dissolved iron of 4.8 and 4.7 mg/L, aluminum of 0.04 and less than 0.01 mg/L, manganese of 0.95 and 0.99 mg/L, and oxygen of 0.4 and 0.2 mg/L, respectively (table 3, fig. 10). The combined load of dissolved iron, aluminum, and manganese at M17 was 11.5 and 3.4 Mg/yr, respectively. The net-alkaline quality of the AMD from the Preston Mine Tunnel indicates aeration alone may be appropriate for its treatment (fig. 2). Land area within Girardville (vacant, abandoned building lots) before the discharge enters Mahanoy Creek could be investigated for a passive aerobic wetland.

The Centralia Mine Tunnel (M19) drains an area of 4 mi^2 (10.4 km^2) near Centralia. Recharge over about half this area originates in the Shamokin Creek Basin. Big Mine Run at S15 is formed almost entirely of AMD discharged at M19. In March and August 2001, discharge at M19 was 3.86 and 2.43 ft^3/s (6,560 and 4,130 L/min) with pH of 3.9 and 3.6, net alkalinity of -59 and -80 mg/L, and concentrations of dissolved iron of 7.6

and 4.6 mg/L, aluminum of 5.5 and 8.7 mg/L, manganese of 4.9 and 5.9 mg/L, and oxygen of 7.4 and 8.4 mg/L, respectively (table 3, fig. 10). In March and August 2001, the combined load of dissolved iron, aluminum, and manganese at M19 was 62.1 and 41.7 Mg/yr, respectively. Less than 10 percent of these metals loads were attenuated prior to the discharge of Big Mine Run (S15) to Mahanoy Creek (table 3, fig. 7). Although other AMD sources in the area discharge larger quantities of metals, the acidity loading from the Centralia Mine Tunnel is larger than that from other AMD sources in the Mahanoy Creek Basin. Passive treatment to add alkalinity and remove iron, aluminum, and other metals may be possible considering the 0.50 mi (0.80 km) distance and relatively large, undeveloped area between the discharge and Mahanoy Creek.

The Bast Mine (M18, M20, and M21) discharges from various mine openings along Mahanoy Creek between Girardville and Ashland. In March and August 2001, discharge from the Bast Mine Tunnel (M18) was 0.67 and 0.40 ft^3/s (1,140 and 680 L/min) with pH of 5.8 and 5.7, net alkalinity of -2 and -24 mg/L, and concentrations of dissolved iron of 7.7 and 19 mg/L, aluminum of 0.35 and 0.47 mg/L, manganese of 2.5 and 4.2 mg/L, and oxygen of 8.3 and 8.8 mg/L, respectively (table 3, fig. 10). In March and August 2001, the combined load of dissolved iron, aluminum, and manganese from M18 was 6.3 and 8.5 Mg/yr, respectively. These data indicate a moderate acidity loading from M18 to Mahanoy Creek but with lower quantities of metals than the net-alkaline AMD at sites M20 and M21. In March 2001, discharge at the Bast Mine overflow (M20) was 2.23 ft^3/s (3,790 L/min) with pH of 6.5, net alkalinity of 80 mg/L, and concentrations of dissolved iron of 8.4 mg/L, aluminum of 0.04 mg/L, manganese of 2.7 mg/L, and oxygen of 3.1 mg/L (table 3, fig. 10). The combined load of dissolved iron, aluminum, and manganese from M20 was 22.2 Mg/yr in March 2001; however, in August 2001, AMD was not flowing at M20. In March and August 2001, discharge at the Bast Mine Oakland Tunnel (M21) was 4.0 and 2.3 ft^3/s (6,800 and 3,910 L/min) with pH of 6.3 and 6.4, net alkalinity of 74 and 79 mg/L, and concentrations of dissolved iron of 7.3 and 14 mg/L, aluminum of 0.05 and 0.08 mg/L, manganese of 2.6 and 3.1 mg/L, and oxygen of 3.8 and 5.2 mg/L, respectively (table 3, fig. 10). During March and August 2001, the combined load of dissolved iron, aluminum, and manganese from M21 was 35.6 and 35.3 Mg/yr, which was among the largest in the Mahanoy Creek Basin. Passive-treatment systems at M18, M20, and M21 would not be suitable because of the proximity of these sites to Mahanoy Creek and roads. Although the discharge from M18 flows through a backwater wetland within the flood plain of Mahanoy Creek, metals removal is not likely to be large given the slightly acidic pH of the discharge.

Four sources of AMD from the Tunnel Mine (M22, M23, M24, and M25) originate from surface drainage and underground mines on the south side of Ashland. Two small seeps at M22 and M23 and two larger discharges from mine openings at M24 and M25 were sampled during this study. In August 2001, seepage from a spoil bank was sampled at M22 with a flow rate of 0.03 ft^3/s (51 L/min), pH of 4.1, net alkalinity of -109 mg/L, and concentrations of dissolved iron of 2.3 mg/L, aluminum of 15 mg/L, manganese of 9.7 mg/L, and oxygen of 5.5 mg/L (table 3). The combined load of dissolved iron, aluminum, and manganese at M22 was 0.8 Mg/yr, mostly because of the high aluminum concentration. In contrast, the Tunnel Mine discharge at M23 in August 2001 had a flow rate of 0.03 ft^3/s (and 51 L/min) with pH of 5.9, net alkalinity of 55 mg/L, and concentrations of dissolved iron of 0.03 mg/L, aluminum of less than 0.01 mg/L, manganese of 2.7 mg/L, and oxygen of 0.1 mg/L (table 3). The combined load of dissolved iron, aluminum, and manganese at M23 was 0.1 Mg/yr. The Tunnel Mine drain pool area and storage (M24) was sampled in March and August 2001 and previously was reported by Sanders and Thomas, Inc. (1975) and Reed and others (1987). In March and August 2001, discharge at M24 was 0.09 and 0.13 ft^3/s (153 and 221 L/min) with pH of 7.4 and 6.1, net alkalinity of 218 and 54 mg/L, and concentrations of dissolved iron of 0.24 and 14 mg/L, aluminum of less than 0.01 and 0.05 mg/L, manganese of 0.82 and 2.7 mg/L, and oxygen of 8.9 and 0.9 mg/L, respectively (table 3). The combined load of dissolved iron, aluminum, and manganese at M24 was 0.1 and 2.0 Mg/yr, respectively. Different results between the sampling dates reflect different flow paths as well as other factors, such as aeration. In March 2001, the AMD was sampled after discharging from the last of three ponds in series, whereas in August 2001, the AMD at that location was not flowing and the sample was collected within the first pond near its outlet. The Tunnel Mine Orchard Drift overflow (M25) was not sampled in March 2001. In August 2001, discharge at M25 was 0.04 ft^3/s (68 L/min) with pH of 5.6, net alkalinity of 21 mg/L, and concentrations of dissolved iron of 0.04 mg/L, aluminum of 0.17 mg/L, manganese of 0.82 mg/L, and oxygen of 1.5 mg/L (table3). The combined load of dissolved iron, aluminum, and manganese at M25 was less than 0.1 Mg/yr. Generally, the seepage with high concentrations of aluminum can be abated by removal of the acid-forming culm bank where it originates, whereas the net-alkaline discharge can be treated by aerobic wetlands.

Two discharges originate from the Potts Mine (M26 and M27) near Locustdale that form the principal source of streamflow to Big Run. In March and August 2001, discharge at the Potts Mine West Breach (M26) was 1.44 and 0.36 ft^3/s (2,450 and 612 L/min) with pH of 6.6 and 6.4, net alkalinity of 245 and 216 mg/L, and concentrations of dissolved iron of 16 and 30 mg/L, aluminum of less than 0.01 and 0.24 mg/L, manganese of 3.6 and 5.1 mg/L, and oxygen of 0.2 and 1.7 mg/L, respectively (table 3). In March and August 2001, the combined load of dissolved iron, aluminum, and manganese was 25.2 and 11.4 Mg/yr, respectively. In March and August 2001, discharge at the Potts Mine East Breach (M27) was 0.22 and 0.29 ft^3/s (374 and 493 L/min) with pH of 6.7 and 6.5, net alkalinity of 190 and 202 mg/L, and concentrations of dissolved iron of 8.9 and 13 mg/L, aluminum of less than 0.01 and 0.01 mg/L, manganese of 2.1 and 2.3 mg/L, and oxygen of 0.6 and 2.5 mg/L, respectively (table 3). In March and August 2001, the combined load of dissolved iron, aluminum, and manganese was 2.2 and 4.0 Mg/yr, respectively. The net-alkaline character of these discharges is remarkable and coupled with a long, turbulent flow path results in more than 80-percent reductions in the iron loading from the AMD sources at M26 and M27 to the mouth of Big Run at S20 (fig. 7). Constructed wetlands could be installed to prevent the iron from accumulating in Big Run where the solids are scoured and transported during high-flow conditions.

The Lavelle Mine slope (M28), Locust Gap Tunnel (M29), Helfenstein Tunnel (M30), and the Doutyville Tunnel (M31) discharge AMD from the Locust Gap Mine and, possibly, the Germantown Mine that drain an area of 7 mi^2 (18.1 km^2) entirely within the Shamokin Creek Basin (Reed and others, 1987). In particular, the Locust Gap Tunnel and Doutyville Tunnel discharges contribute substantial quantities of AMD to Mahanoy Creek where the stream traverses an extensively forested area that is not scarred by mining or underlain by anthracite-bearing rocks. In March and August 2001, discharge at the Lavelle Mine slope (M28) near Mowry was 0.23 and 0.01 ft^3/s (391 and 17 L/min) with pH of 3.9 and 4.3, net alkalinity of -20 and -32 mg/L, and concentrations of dissolved iron of 2.2 and 10 mg/L, aluminum of 1.3 and 1.5 mg/L, manganese of 1.6 and 1.8 mg/L, and oxygen of 0.5 and 1.0 mg/L, respectively (table 3). The combined load of dissolved iron, aluminum, and manganese was 1.0 and 0.1 Mg/yr, respectively. Passive treatment of this net acidic but relatively small AMD source could be possible with an anoxic limestone drain and aerobic wetlands given the large undeveloped area between the discharge source and Mahanoy Creek. However, this AMD source with a relatively small metals loading may not warrant priority consideration.

The Locust Gap Tunnel (M29) and Helfenstein Tunnel (M30), both near the village of Helfenstein but 0.6 mi (0.9 km) apart (fig. 1), mistakenly have been described with the same name. However, the drainage from the Locust Gap Tunnel (M29) at the bank of Mahanoy Creek is a large flow compared to the small seep from the hillside at the original Helfenstein Tunnel (M30). In March and August 2001, discharge at the Locust Gap Tunnel (M29) was 17.2 and 7.29 ft^3/s (29,200 and 12,400 L/min) with pH of 6.7 and 6.5, net alkalinity of 34 and 29 mg/L, and concentrations of dissolved iron of 6.7 and 11 mg/L, aluminum of 0.05 mg/L, manganese of 2.3 and 3.8 mg/L, and oxygen of 8.8 and 9.1 mg/L, respectively (table 3, fig. 5). The combined load of dissolved iron, aluminum, and manganese at M29 was 139 and 96.7 Mg/yr, distinguishing the Locust Gap Tunnel as the second largest, continuously flowing AMD source in the Mahanoy Creek Basin after the Packer #5 Mine discharges (M12 and M13). In March and August 2001, the Helfenstein Tunnel (M30) discharged 0.09 and less than 0.01 ft^3/s (153 and 2 L/min) with a combined load of dissolved iron, aluminum, and manganese of less than 0.1 Mg/yr (table 3, fig. 5). Because of its small size and remote location, the discharge at M30 may not warrant treatment, even though land may be available for a treatment system. In contrast, passive treatment of the discharge from the Locust Gap Tunnel (M29) would not be suitable because of its large size and proximity to Mahanoy Creek. Although the metals discharged are substantial from the Locust Gap Tunnel (M29), this AMD source may be considered low priority for cleanup on the basis of its marginally net alkaline character (table 3). Furthermore, the aquatic ecosystem downstream of M29 on Mahanoy Creek near Gowen City (S23) is relatively high quality compared to upstream segments (tables 4 and 5).

The Doutyville Tunnel (M31) flows into Mahanoy Creek 2.6 mi (4.2 km) downstream from the Locust Gap Tunnel (M29) and 4.4 mi (7.1 km) upstream from site S23 on Mahanoy Creek (fig. 1). In March and August 2001, discharge at M31 was 3.52 and 0.99 ft^3/s (5,980 and 1,680 L/min) with pH of 5.0 and 6.1, net alkalinity of -17 and 9 mg/L, and concentrations of dissolved iron of 2.1 and 3.4 mg/L, aluminum of 1.8 and 0.34 mg/L, manganese of 1.3 and 1.5 mg/L, and oxygen of 9.3 and 9.6 mg/L, respectively (table 3, fig. 5). In March and August 2001, the combined load of dissolved iron, aluminum, and manganese at M31 was 16.4 and 4.6 Mg/yr, respectively. Passive treatment of the discharge from the Doutyville Tunnel (M31) may not be suitable because of its large size, remote location, proximity to Mahanoy Creek, and its variable flow and water quality (table 3). This AMD source also may be considered low priority for cleanup because of the relatively high-quality aquatic ecosystem downstream on Mahanoy Creek near Gowen City (S23) (tables 4 and 5).

The North Franklin Mine, which underlies a mined area of 3 mi^2 (4.8 km^2) in the upper part of Zerbe Run near Trevorton, discharges AMD from at least four continuous and intermittent sources (M32, M33, M34, and M35) (fig. 1). The largest AMD source in the Zerbe Run subbasin is the North Franklin Mine Drift and Borehole (M32) that accounts for most of the flow and metals loading to the unnamed tributary to Zerbe Run at S25 (figs. 1, 9, and 10). In March and August 2001, discharge at M32 was 6.45 and 2.56 ft^3/s (11,000 and 4,350 L/min) with relatively constant pH of 5.9 and 5.2, net alkalinity of 5 and 3 mg/L, and concentrations of dissolved iron of 11 and 13 mg/L, aluminum of 0.32 and 0.51 mg/L, manganese of 1.9 and 2.3 mg/L, and oxygen of 4.6 and 4.4 mg/L, respectively (table 3, fig. 5). In March and August 2001, the combined load of dissolved iron, aluminum, and manganese at M32 was 76.2 and 36.2 Mg/yr, respectively, ranking it among the top five sources of AMD in the Mahanoy Creek Basin. Discharge at the North Franklin Mine seepage (M33) in March 2001 was 0.03 ft^3/s (51 L/min) with pH of 3.1, net alkalinity of -110 mg/L, and concentrations of dissolved iron of 5.1 mg/L, aluminum of 9.8 mg/L, manganese of 3.8 mg/L, and oxygen of 4.8 mg/L (table 3, fig. 5). The load of dissolved iron, aluminum, and manganese at M33 was 0.5 Mg/yr in March 2001; however, in August 2001, site M33 was dry. In March and August 2001, seepage from a culm bank at M34 had a flow rate of 0.02 and less than 0.01 ft^3/s (34 and less than 2 L/min) with pH of 2.9, net alkalinity of -184 and -220 mg/L, and concentrations of dissolved iron of 21 and 31 mg/L, aluminum of 14 and 17 mg/L, manganese of 3 and 3.9 mg/L, and oxygen of 0.9 and 0.6 mg/L, respectively (table 3, fig. 5). The pH was lower and concentrations of dissolved aluminum were greater than those for any other AMD source sampled during the study. Despite the small flow rate, the combined load of dissolved iron, aluminum, and manganese at M34 was 0.7 and 0.1 Mg/yr in March and August 2001, respectively. The Sunshine Mine overflow (M35) discharge at M35 was not sampled because a locked gate blocked access to the AMD source; discharge was not observed flowing from the site. Passive-treatment wetlands may be suitable for treatment of the North Franklin Drift and Borehole at M32 because it contributes the greatest loading of metals to Zerbe Run and it has marginally net alkaline water quality. Nevertheless, other smaller AMD sources could cause acidification of Zerbe Run that was apparent at S26 during low base-flow conditions (table 3). Ideally, alkalinity may be added to the unnamed tributary of Zerbe Run in a sufficient quantity to buffer downstream loading of AMD.

Remedial Priorities and Alternatives

Flow and concentration data for the high base-flow samples collected in March 2001 were used to determine priority ranks of the AMD sources on the basis of loads of dissolved iron, manganese, and aluminum and to indicate the minimum size of wetlands for iron removal. The AMD source with the highest loading was assigned a rank of 1, with successively higher ranks assigned to AMD sources in descending order of dissolved metal loading (table 8). To provide context for comparing the AMD sources, the dissolved metals loading at each AMD source was expressed as a percentage of the sum of dissolved metals loading for all sampled AMD sources in the watershed (table 8). Generally, the AMD sources with the largest flow rates and iron concentrations were ranked among the top 15 AMD sources; however, the AMD ranking generally did not correlate with the acidity, pH, or aluminum concentration (fig. 11). Although concentrations increased with decreased flow (fig. 3), the contaminant loadings generally increased with flow.

The top 4 AMD sources, Locust Gap Tunnel (M29), Packer #5 Breach (M13), Packer #5 Borehole (M12), and Girard Mine seepage (M11), on the basis of dissolved metals loading in March 2001 accounted for more than 50 percent of the metals loading to Mahanoy Creek, whereas the top 15 AMD sources accounted for more than 99 percent of the metals loading (table 8). When sampled in March 2001, the top 15 AMD sources had flow rates ranging from 0.4 to 17.2 ft^3/s (680 to 29,200 L/min) and pH from 3.9 to 6.7. Nine of the top 15 AMD sources, including the top 4, were net alkaline (alkalinity greater than acidity); the others were net acidic and will require additional alkalinity to facilitate metals removal and maintain near-neutral pH.

The March 2001 high base-flow data for flow rate and dissolved metal concentrations were considered useful in the evaluation of AMD priorities because (1) flow rates in March 2001 were near normal based on long-term streamflow record for Shamokin Creek (Cravotta and Kirby, 2004a), (2) six previously identified intermittent AMD sources were not discharging during the August 2001 low base-flow survey, and (3) acidity is determined largely by dissolved metals concentrations (Cravotta and Kirby, 2004b). Ideally, loadings and associated AMD priorities should be determined on the basis of long-term averages, but these data were not available. Data for pH were not used for the ranking computations because pH tends to be an unstable parameter that does not indicate the ultimate potential for acidic conditions (Cravotta and Kirby, 2004b). Furthermore, when pH or hydrogen ion loadings were included in the ranking computations, results were not changed appreciably. Estimates of the metals loads and corresponding rankings of AMD priorities also were similar on the basis of the metals in whole-water (total) and 0.45-μm filtered (dissolved) subsamples.

The ranking sequence for the top AMD sources based on the high base-flow data generally matched that based on the low base-flow data (fig. 12). However, 2 of the top 15 AMD sources, the Vulcan-Buck Mountain seepage (M02) and the Bast Mine overflow (M20), ranked 7 and 11, respectively, were not flowing in August 2001 (table 3, fig. 12). With the exception of AMD sources with elevated concentrations of aluminum, such as the Vulcan-Buck Mountain Mine (M02 and M03), Centralia Mine (M19), and Doutyville Tunnel (M31), the concentration of dissolved iron greatly exceeded the other metals, indicating iron was the predominant source of acidity (fig. 12). Manganese typically was greater than or equal to the aluminum concentration.

The AMD priority ranking could have been developed using various other constituents or computational methods. Because the proportions of dissolved iron, aluminum, and manganese in the AMD varied from site to site, different rankings could result by weighting the metals with different factors such as dividing the concentration by regulatory standards. Cherry and others (2001) and Herlihy and others (1990) used a combination of biological and chemical metrics to assess AMD effects on a watershed scale. Williams and others (1996, 1999) used flow and chemical constituents including acidity, metals, and sulfate to develop a ranking scheme based primarily on contaminant loading; pH was used as a “tie-breaker.” For the current study, rankings on the basis of sulfate were similar to those computed on the basis of dissolved metals (table 8). When net-alkalinity loading was considered, the ranks for various AMD sources with substantial alkalinity and metals loading shifted to lower ranks (table 8). For example, the top five AMD sources on the basis of metals loading, Locust Gap Tunnel (M29), Packer #5 Mine Breach (M13) and Borehole (M12), Girard Mine seepage (M11), and North Franklin Mine Drift and Borehole (M32), had net-alkalinity rankings of 25, 20, 24, 21, and 15, respectively (table 8). These rankings indicate that acidity loading from these sources is less than that from other top-ranked AMD sources; however, because of site specific limitations, their treatment is not necessarily more feasible than other large AMD sources. Ultimately, the feasibility of remediation of a particular discharge must consider the AMD quality and loading rates, if the site is accessible for treatment, and if funding, construction permits, and other resources can be obtained for implementation. Although such details have not been considered for this assessment, possible remedial alternatives and comments on site-specific issues for consideration by resource

Table 8. Priority rankings and possible remedial alternatives for abandoned mine drainage in Mahanoy Creek Basin, Pennsylvania.

[Priority rankings based on instantaneous loadings of dissolved metals, net alkalinity, or sulfate during March 26-28, 2001. Remedial alternatives are not identified in order of preference; any treatment design would require additional data and specific analysis; VFCW, vertical-flow compost wetland; ALD, anoxic limestone drain; OLD, flushable oxic limestone drain; OLC, open limestone channel; X, applicable; +, additional; -, not applicable; ?, insufficient data; =, equal to; >, greater than; ≥, greater than or equal to; <, less than; ≤, less than or equal to; g/m^2/d, grams per meter squared per day; lb/acre/d, pounds per acre per day]

Site name and local identification number[1]	Percentage dissolved Fe, Al, and Mn load	Metals rank[2]	Net alkalinity rank	Sulfate rank	Principal characteristics[3]	Remedial Alternatives[4]							Comments	Wetland area, acres[5]
						Remove culm bank	VFCW	ALD	OLD	OLC	Aerobic pond(s)	Active treatment		
Locust Gap Mine Locust Gap Tunnel (M29)	18.7	1	25	1	Very large flow; high Fe, Mn; low Al; net alkaline; oxic	-	-	-	-	-	X	X	Area inadequate for passive treatment.	5.8
Packer #5 Mine Breach (M13)	11.6	2	20	3	Very large flow; very high Fe, Mn; low Al; net alkaline; anoxic	-	-	-	-	-	X	X	Area adequate for wetland (?). Combine w/borehole.	3.9
Packer #5 Mine Borehole (M12)	11.4	3	24	2	Very large flow; high Fe, Mn; low Al; net alkaline; anoxic	-	-	-	-	-	X	X	Area adequate for wetland (?). Combine w/breach.	3.4
Girard Mine seepage (M11)	10.8	4	21	9	Very large flow; high Fe, Mn; low Al; net alkaline; oxic	-	-	-	-	-	X	X	Area inadequate for passive treatment.	3.0
N. Franklin Mine Drift and Borehole (M32)	10.3	5	15	4	Very large flow; moderate Fe, Mn; moderate Al; net alkaline?; oxic	-	-	-	X	-	+	X	Area available for passive treatment.	2.6
Centralia Mine Tunnel (M19)	8.4	6	1	5	Very large flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	X	Area available for passive treatment.	0.9
Vulcan-Buck Mtn. Mine seepage (M02)	7.1	7	2	8	Intermittent flow; moderate Fe, Mn; high Al; net acidic; anoxic	-	-	-	-	X	-	-	Fill channel with limestone or consider diversion well.	0.9
Bast Mine Oakland Tunnel (M21)	4.8	8	22	6	Very large flow; high Fe, Mn; low Al; net alkaline; oxic	-	-	-	-	-	X	X	Area inadequate for passive treatment.	1.7
Vulcan-Buck Mtn. Mine Boreholes (M03)	4.1	9	3	12	Very large flow; moderate Fe, Mn; moderate Al; net acidic; anoxic	-	-	X	-	X	+	X	Area inadequate for passive treatment.	1.0
Potts Mine West Breach (M26)	3.4	10	23	7	Large flow; high Fe, Mn; moderate Al; net alkaline; suboxic	-	-	-	-	-	X	-	Area adequate for wetland (?).	1.3
Bast Mine Overflow (M20)	3.0	11	19	10	Intermittent flow; high Fe, Mn; low Al; net alkaline; oxic	-	-	-	-	-	X	-	Area adequate for wetland (?).	0.6
Locust Gap Mine Doutyville Tunnel (M31)	2.2	12	4	11	Large flow; moderate Fe, Mn; moderate Al; net acidic?; oxic	-	-	-	X	X	+	-	Area inadequate for passive treatment (?).	0.4
Preston Mine #3 Tunnel overflow (M17)	1.6	13	18	15	Large flow; moderate Fe, Mn; low Al; net alkaline; anoxic	-	-	-	-	-	X	-	Area inadequate for passive treatment (?).	0.3
Weston Mine Lost Cr Borehole (M07)	0.9	14	14	14	Moderate flow; very high Fe, Mn; low Al; net alkaline; anoxic	-	-	-	-	-	X	-	Discharge drains to existing wetland.	0.4
Bast Mine Tunnel (M18)	0.9	15	9	13	Moderate flow; high Fe, Mn; moderate Al; net acidic; oxic	-	-	-	X	-	+	-	Area available for passive treatment.	0.4
Potts Mine East Breach (M27)	0.3	16	16	17	Moderate flow; high Fe, Mn; low Al; net alkaline; oxic	-	-	-	-	-	X	-	Area adequate for wetland (?).	0.1
Hammond Mine Seepage (M08)	0.3	17	17	16	Intermittent flow; moderate Fe, Mn; low Al; net alkaline; anoxic	-	-	-	-	-	X	-	Discharge drains to existing wetland.	<.1
Lavelle Mine Lavelle slope (M28)	0.1	18	5	19	Moderate flow; moderate Fe, Mn; moderate Al; net acidic; anoxic	-	-	X	-	-	+	-	Area available for passive treatment.	0.1
Tunnel Mine seepage to ditch (M22)	0.1	19	7	18	Small flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	-	Source removal best; area available for treatment.	<.1
N. Franklin Mine bank seepage (M34)	0.1	20	6	22	Small flow; high Fe, Mn; very high Al; net acidic; anoxic	X	-	-	-	-	-	-	Source removal best; area available for treatment.	<.1
N. Franklin Mine seepage (M33)	0.1	21	8	21	Intermittent flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	-	Source removal best; area available for treatment.	<.1
Tunnel Mine drain pool area (M24)	<0.1	22	13	20	Moderate flow; moderate Fe, Mn; low Al; net alkaline; oxic	-	-	-	-	-	X	-	Possibly enhance existing ponds for treatment.	0.1
Tunnel Mine discharge from spoil bank (M23)	<0.1	23	12	23	Small flow; moderate Fe, Mn; low Al; net alkaline; anoxic	-	-	-	-	-	X	-	Area available for passive treatment.	<.1
Tunnel Mine Orchard Drift overflow (M25)	<0.1	24	11	24	Small flow; low Fe, Mn; low Al; net alkaline; suboxic	-	-	-	-	-	X	-	Area available for passive treatment.	<.1
Locust Gap Mine Helfenstein Tunnel (M30)	<0.1	25	10	25	Small flow; low Fe, Mn; low Al; net alkaline; oxic	-	-	-	-	-	X	-	Discharge drains to existing wetland.	<.1
Weston Mine surface areas seepage (M05)	<0.1	26	28	26	Intermittent flow; moderate Fe, Mn; moderate Al; net alkaline?; oxic	?	-	-	-	-	-	-	Insufficient data.	-
Vulcan-Buck Mtn. Mine Morris Tunnel (M01)	<0.1	27	26	27	Intermittent flow; not sampled.	?	-	-	-	-	-	-	Insufficient data.	-
Gilberton Mine Pump (M04)	<0.1	28	27	28	Intermittent very large flow; not sampled.	-	-	-	-	-	-	?	Insufficient data.	-

1. Site descriptions given in tables 1 and 3.

2. Rankings based on instantaneous loadings computed as product of flow rate and concentration of relevant constituents. Metals loading based on concentrations of dissolved iron, aluminum, and manganese. Net-alkalinity concentration computed as measured alkalinity minus computed acidity per equations 7 and 8. Rank of 1 for greatest loading; rank value increases with decreased loading.

3. Principal characteristics based on maxima and minima for flow rate and concentrations of alkalinity, dissolved metals, and oxygen (in mg/L) for samples collected in March and August 2001 (table 3). Flow (ft^3/s): 'very large' if minimum > 2.0; 'large' if maximum > 1.0 and ≤ 2.0; 'moderate' if maximum > 0.1 and ≤ 1.0; 'small' if maximum ≤ 0.1; 'intermittent' if maximum or minimum = 0. Iron and manganese (mg/L): 'very high' if minimum Fe > 12 and minimum Mn > 4; 'high' if minimum Fe ≤ 12 and minimum Mn ≤ 4; 'moderate' if minimum Fe ≤ 6 and minimum Mn ≤ 2; 'low' if maximum Fe < 3 and maximum Mn < 1. Aluminum (mg/L): 'very high' if maximum > 4; 'high' if maximum > 2; 'moderate' if maximum ≤ 2; 'low' if maximum < 0.2. Net alkalinity (alkalinity - computed acidity; mg/L as CaCO$_3$): 'net acidic?' if maximum ≤ 5; 'net acidic' if maximum ≤ 0; 'net alkaline?' if minimum > 0 or if missing and minimum pH ≥ 6.4; 'net alkaline' if minimum ≥ 5. Dissolved oxygen (mg/L): 'anoxic' if maximum ≤ 1; 'suboxic' if maximum ≤ 2; 'oxic' if minimum > 2.

4. Remedial alternatives initially identified on the basis of maxima and minima for flow rate and water quality (in mg/L): 'Remove culm bank' if maximum pH < 4.0; 'Aerobic pond' if minimum net alkalinity ≥ 5; 'VFCW and aerobic pond' if minimum net alkalinity < 5, maximum dissolved oxygen > 1, maximum Al ≥ 2, and maximum flow ≤ 6.5; 'ALD and aerobic pond' if minimum net alkalinity < 5, maximum dissolved oxygen ≤ 1, maximum Al < 2, and maximum flow ≤ 6.5; 'OLD and aerobic pond' if minimum net alkalinity < 5, maximum dissolved oxygen > 1, maximum Al < 3, and maximum flow ≤ 6.5; 'OLC' if minimum net alkalinity < 5, maximum Fe < 10, maximum Al < 5, and maximum flow ≤ 10; 'Active Treatment' if minimum flow > 2 or maximum net alkalinity < -300.

5. Minimum wetland size computed by dividing the product of maximum flow rate and maximum iron concentration, in grams per day, by 20 g/m^2/d (180 lb/acre/d) per Hedin and others (1994). If smaller loading rate used, increase area by constant factor: for loading rate of 10 or 5 g/m^2/d, multiply wetland area estimate by 2 or 4, respectively.

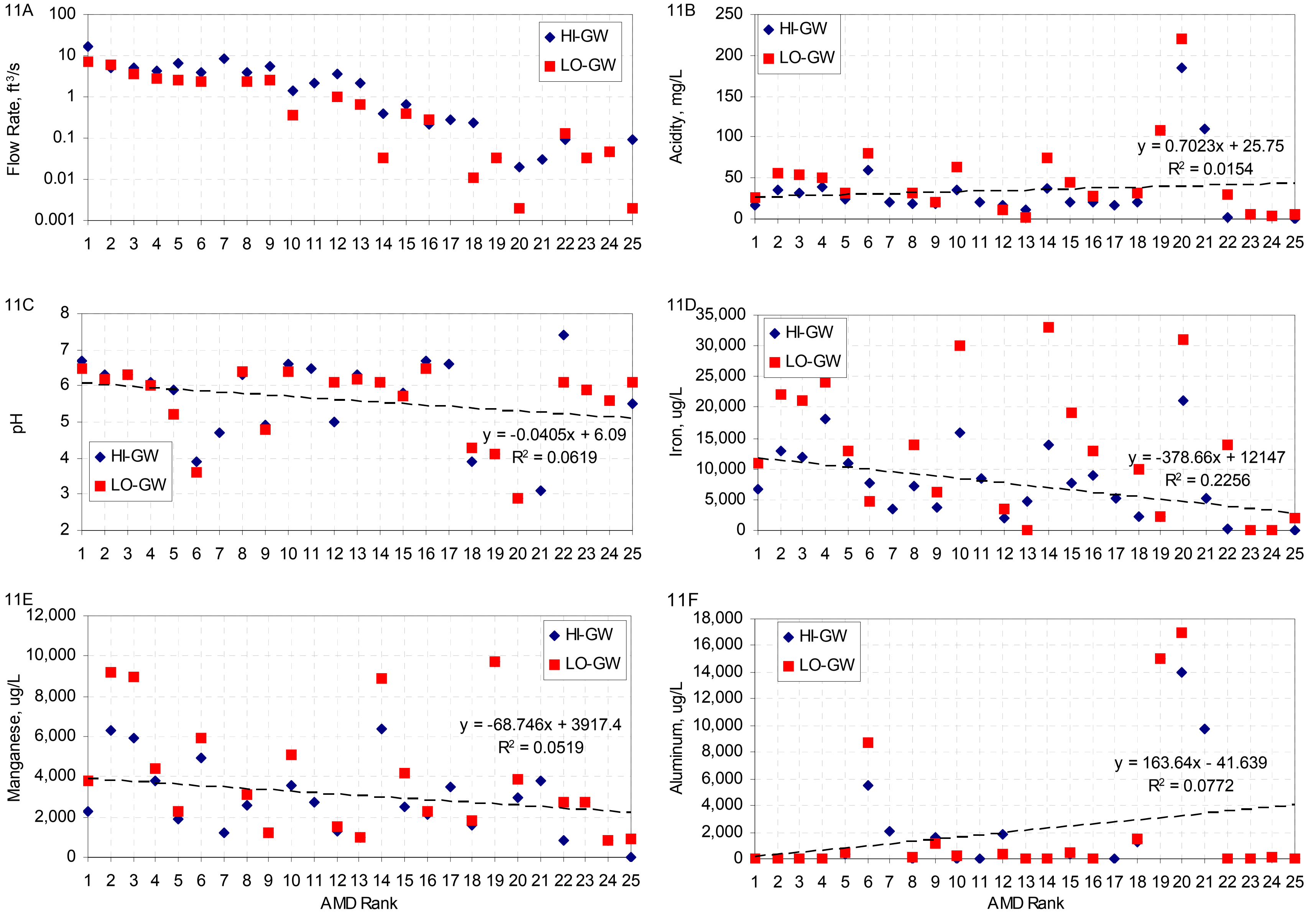

Figure 11. Relation between priority ranking based on dissolved metals loading in March 2001 for top 25 abandoned mine drainage (AMD) sites and (A) flow rate, (B) acidity, (C) pH, (D) iron, (E) manganese, and (F) aluminum concentrations for high base-flow (HI-GW) and low base-flow (LO-GW) AMD samples, Mahanoy Creek Basin, Pennsylvania. Flow rate in cubic feet per second (ft^3/s); concentrations in millgrams per liter (mg/L) or micrograms per liter (μg/L).

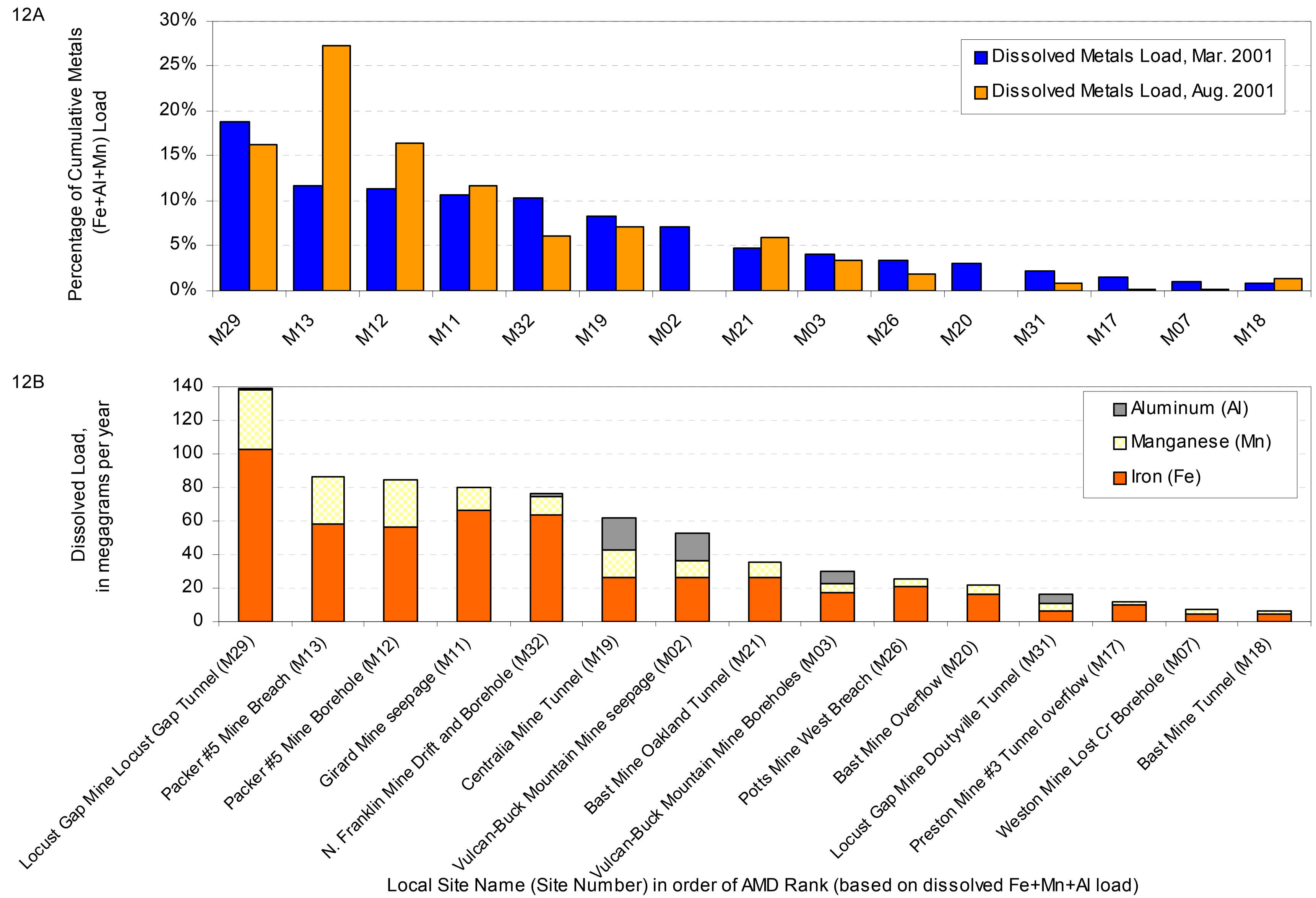

Figure 12. Comparison of priority ranks for top 15 abandoned mine drainage (AMD) sites, Mahanoy Creek Basin, Pennsylvania, (A) on the basis of concentrations of iron, aluminum, and manganese in filtered samples collected during high base-flow conditions (HI-GW) in March 2001 and low base-flow conditions (LO-GW) in August 2001, and (B) considering relative contributions of dissolved iron, manganese, and aluminum to the dissolved metals loading during March 2001.

managers and land owners that may be involved in decisions to implement remediation are summarized in table 8.

Generally, to meet water-quality criteria for 0.3 mg/L dissolved iron, nearly all the AMD sources would require construction of some sort of settling basin or wetland to facilitate iron oxidation, hydrolysis, and deposition. Hence, to provide a basis for evaluating alternatives for passive treatment, the minimum wetland size for each AMD source was computed using the data for maximum flow rate and maximum iron concentration for the March 2001 and August 2001 data and considering criteria of Hedin and others (1994) for an iron-removal rate of 180 lb/acre/d (20 g/m^2/d) (table 8). The computed wetland sizes ranged from 5.8 acres for the Locust Gap Tunnel discharge (M29) to less than 0.1 acre for seven small AMD discharges. Small wetland acreages were computed for sites with low flow rates and low concentrations of dissolved iron; however, many of these AMD sources, such as seepage from the North Franklin Mine (M33 and M34) or the Tunnel Mine (M22), could have high concentrations of dissolved aluminum (table 8). Consequently, a larger treatment area than that computed based on iron alone may be needed.

If the AMD is net acidic and (or) has elevated concentrations of aluminum, treatment steps or components that add alkalinity to the AMD could be appropriate in addition to a wetland (fig. 2). Because many of the AMD sources in the Mahanoy Creek Basin have large flow and metal loading rates (table 8), innovative designs that accelerate iron oxidation (Dietz and Dempsey, 2002) and (or) incorporate automatic flushing for solids removal (Vinci and Schmidt, 2001; Weaver and others, 2004; Schueck and others, 2004) may be advantageous. Furthermore, bench-scale testing of the possible treatment alternatives, such as that by Cravotta (2002, 2003), Cravotta and others (2004), and Dietz and Dempsey (2002), could be helpful for the selection and design of treatment alternatives.

Various restoration activities could be considered to mitigate the AMD contamination in the Mahanoy Creek Basin. Because many of the AMD sources are large or have insufficient land area for construction of active or passive-treatment systems, the prevention of infiltration through mine spoil or into the underground mines is warranted. If surface reclamation or streamflow restoration is planned or completed, the design of any AMD treatment system should consider additional monitoring to document potential changes in flow and loading rates. The following restoration strategies that were identified to meet TMDLs in the Shamokin Creek Basin (Pennsylvania Department of Environmental Protection, 2001; Cravotta and Kirby, 2004a) generally could be applicable in the Mahanoy Creek Basin and other watersheds affected by abandoned mines.

- Reclamation of abandoned surface mines, including removal of abandoned highwalls and spoil banks and filling abandoned surface-mine pits would eliminate surface-water accumulations that become contaminated with mine drainage because of contact with exposed acid-producing strata and reduce the amount of surface runoff directed into the mine-pool systems. The regrading of disturbed areas, if returned to original contour before mining, would provide a more natural flow pattern for runoff and prevent surface water from percolating through abandoned refuse and entering underground mine pools.
- Removal, regrading, and (or) replanting of abandoned coal-refuse piles would reduce the amount of sediments, silt, and coal-waste runoff into surface streams and eliminate a source of AMD.
- Restoration of surface channels and flow of streams that now disappear into spoil banks and enter deep-mine pools could lessen the volume of water discharged by AMD sources.
- Site-specific assessments to determine whether passive treatment is practical and which treatment systems are best suited for specific discharges should include discharge water quality and flow, topographical setting, construction costs, and long-term operation and maintenance costs. Suitable technology may not be available to passively treat many of these high-volume discharges.

Data Usage and Limitations

The evaluation of contaminant loading rates and the use of these data for the development of TMDLs or design of treatment systems require sufficient samples to characterize the average values and extremes in flow rates and quality at each of the AMD sources. However, average or extreme conditions generally are not known for individual AMD or stream monitoring sites in the Mahanoy Creek Basin because continuous records for flow or chemistry are not available. Detailed treatment design at any AMD site would require additional data on the range of flow rates and corresponding variations in water quality.

In the Mahanoy Creek Basin, the primary water-quality effects from AMD are locally low pH, high concentrations of aluminum and possibly other toxic metals, and extensive iron-hydroxide coatings on streambeds. Low pH coupled with high concentrations of dissolved aluminum and other metals, such as nickel and zinc, can be toxic to many aquatic organisms (Burrows, 1977; Burton and Allan, 1986; Hyman and Watzlaf, 1997; Earle and Callaghan, 1998; U.S. Environmental Protection Agency, 2002a). Although dissolved iron is not acutely toxic at the concentrations documented in this basin (less than 50 mg/L), thick accumulations of iron hydroxide effectively can eliminate the habitat of aquatic macroinvertebrates that serve as food sources for organisms at higher trophic levels. Manganese loading was incorporated in the AMD prioritization because manganese concentrations in streamwater and in effluent from active mines are regulated by Federal and State law (Commonwealth of Pennsylvania, 1998a, 1998b, 2001a). Kleinmann and Watzlaf (1986) and Hyman and Watzlaf (1997) explained that the manganese regulation is in force not because of its toxicity, but rather because manganese concentration serves as a proxy for toxic trace metals. The streambed sediments collected for this study indicated correlations between manganese and cadmium, cobalt, nickel, strontium, and zinc.

An inherent assumption in the three-metal assessment used for this study is that, if the instream loadings of iron, manganese, and aluminum are decreased to acceptable levels by appropriate treatment or natural processes, other chemical, physical, and biological parameters will be within a range that should support normal aquatic populations. For pH, acidity, and alkalinity, this assumption should be valid because the AMD treatments appropriate for removal of iron, manganese, and aluminum also will increase the pH and alkalinity while decreasing acidity in receiving streams. Because trace-metal concentrations in streamwater and sediments tend to be controlled by their adsorption to precipitated Fe(III) and Mn(IV) hydroxides, trace-metal toxicity also can be reduced by treatments that effectively remove acidity and promote the formation of the metal hydroxides. Dissolved oxygen concentrations in flowing streams should be adequate to support warm-water and (or) cold-water fish populations once the chemical-oxygen demand associated with Fe(II) and Mn(II) loadings is reduced. Although sulfate concentrations will be slightly affected by AMD treatment, the sulfate concentrations in this basin are not expected to negatively affect biological recovery.

Once streamwater chemistry improves, some obstacles may still remain in the restoration of aquatic ecosystems. One remaining obstacle could be lack of adequate aquatic habitat or stream substrate. Most stream substrates in the Mahanoy Creek Basin are coated by relatively loosely bound iron-hydroxide precipitate or they are “armored” by tightly bound iron-hydroxide cements. The loosely bound precipitate can be scoured and suspended by fast-flowing water during stormflow conditions producing turbid, rust-colored streamwater. These conditions are evident during stormflow conditions on Mahanoy Creek, Shamokin Creek (Cravotta and Kirby, 2004a), and Swatara Creek (Cravotta and Bilger, 2001). At least two major remaining substrate or habitat problems will not be addressed by AMD treatment alone. First, during high-flow events, the erosion of coal and waste-rock particles from numerous mine roads and waste-rock piles into streams periodically will disturb stream habitats. Second, some stream reaches (especially the headwaters of Mahanoy Creek and the lower reaches of North Mahanoy Creek, Waste House Run, and Lost Creek) are ephemeral (always dry except during high base-flow or stormflow events) because water is lost to underlying deep mine complexes. The three-metal dissolved load ranking for base-flow conditions does not address these problems.

Although cost estimates were not determined for the remedial alternatives, the flow and water-quality data collected for this study can be used to estimate and compare the relative sizes and costs (capital and annual maintenance) for different active-treatment and passive-treatment alternatives. The flow, acidity, alkalinity, and metals concentration data are critical data for such computations because they determine the quantities of neutralizing agents, the overall size of the treatment system, and the quantities of sludge that may require disposal. Given the estimated size and cost for a treatment system, its feasibility can be evaluated based on analysis with a computer program such as AMDTreat (U.S. Office of Surface Mining Reclamation and Enforcement, 2002). The comparison of alternatives becomes complicated, however, considering different assumptions about capital and long-term maintenance costs, replacement frequency, inflation rates, and interest growth. For example, as shown in appendix D, maximum values for flow, acidity, and metals concentration data collected in March and August 2001 for the Centralia Tunnel discharge (M19) were used with AMDTreat version 3.1 and default settings for unit costs (U.S. Office of Surface Mining Reclamation and Enforcement, 2002). Active treatment of M19 with pebble lime was estimated to have a capital cost of approximately $96,000 and an annual maintenance cost including chemicals of approximately $69,400. Alternatively, a passive system, such as a vertical flow compost wetland (VFCW), is estimated to have a capital cost of approximately $353,000 and annual maintenance costs of approximately $20,400. For a design life of 20 years, annual inflation rate of 3 percent, and annual interest growth rate of 2.5 percent, analysis with AMDTreat indicates that the net present cost for treatment of the Centralia Mine discharge would be approximately $1,014,000 for the active system compared to $668,000 for the passive system. If inflation remained at 3 percent and the interest growth rate were 9.4 percent or more, the net present costs for the active treatments would be less than that for passive treatment. This example illustrates “high capital, but low maintenance cost” for passive treatment compared to active treatment of large discharges and how these costs are considered in future planning. Nevertheless, land for construction of a passive system may not be available or accessible, and the best treatment alternative may not have been considered. Any treatment design would require such site-specific information.

Summary

Mahanoy Creek is a tributary of the Susquehanna River in east-central Pennsylvania. The Mahanoy Creek Basin encompasses an area of 157 mi^2 (407 km^2) including approximately 42 mi^2 (109 km^2) underlain by the Western Middle Anthracite Field. Contaminated runoff and discharges from abandoned anthracite mines degrade the aquatic ecosystem and water quality of the main stem of Mahanoy Creek from its headwaters to its mouth on the Susquehanna River plus various tributaries, including North Mahanoy Creek, Waste House Run, Shenandoah Creek, Zerbe Run, and two unnamed tributaries locally called Big Mine Run and Big Run. The Little Mahanoy Creek and Schwaben Creek are the only major tributaries not affected by mining. To assess the effects of AMD sources and to identify possible remedial alternatives, the USGS, in cooperation with the PaDEP, SCD, and MCWA began a study in 2001. Data on the flow rate and quality of water were collected at all known AMD sites and at selected stream sites within the Mahanoy Creek Basin during high base-flow conditions in March 2001 and then repeated during low base-flow conditions in August 2001. The basin-wide synoptic monitoring of flow and water quality during stable base-flow conditions was performed to (1) identify site-specific characteristics including temporal variability associated with seasonal changes in base flow, (2) indicate spatial variability and relative effects of the AMD throughout the basin, and (3) avoid complications in data collection and interpretation associated with rainfall or other short-duration weather events. Streambed sediments were collected during the August 2001 low base-flow survey. Additionally, during October 2001, data on the diversity and biomass of fish species were collected at a subset of the sampled stream sites, and during 2000-2002, data on the diversity of aquatic macroinvertebrates were collected throughout the basin.

The quantity and quality of water in Mahanoy Creek and tributaries in the upper Mahanoy Creek Basin above the borough of Ashland are affected by leakage to abandoned underground mines and by metal-contaminated discharges from tunnels, slopes, shafts, and unreclaimed spoil associated with abandoned mines. Additional mine discharges affect the quality of water below Ashland, including large discharges from the Locust Gap and Doutyville Tunnels near Helfenstein. High base-flow samples collected in March 2001 and low base-flow samples collected in August 2001 provided information on the current water-quality characteristics and the relative differences among AMD sources and stream-sampling sites. Because as much as one-sixth of the known AMD sources were not flowing during August 2001, the data for March 2001 were used as the primary basis for characterizing aquatic quality at each site and distinguishing AMD priorities. Generally, concentrations of sulfate, iron, and manganese for a particular AMD site were similar or greater during low base-flow than high base-flow conditions.

The pH of most of the AMD sources was near neutral (pH 6 to 7). However, some of the AMD sources with near-neutral and lower pH values had negative net alkalinity, indicating that pH ultimately could decline to values less than 4.5 after complete oxidation and hydrolysis of dissolved metals. The predominant source of acidity in most streamwater samples was dissolved iron. Some streamwater and AMD samples, mainly in the headwaters of Mahanoy Creek, the Big Mine Run drainage from the Centralia Tunnel, and seepage from culm banks of the Tunnel Mine along Mahanoy Creek and the North Franklin Mine, had low pH and elevated concentrations of aluminum (greater than 1 mg/L).

Twenty species of fish were identified in Schwaben Creek near Red Cross, which drains an unmined area of 22.7 mi^2 (58.8 km^2) in the lower part of the Mahanoy Creek Basin. In contrast, 14 species of fish were identified in Mahanoy Creek near its mouth at Kneass, below Schwaben Creek. The diversity and abundance of fish species in Mahanoy Creek decreased progressively upstream from 13 species at Gowen City to only 2 species each at Ashland and Girardville. White sucker (*Catostomus commersoni*), a pollution-tolerant species, was present at each of the surveyed reaches. The presence of fish at Girardville was unexpected because of the poor water quality and iron-encrusted streambed at this location. Generally, macroinvertebrate diversity and abundance at these sites were diminished compared to Schwaben Creek and other unmined tributaries, consistent with the observed quality of streamwater.

A majority of the base-flow streamwater samples met water-quality standards for pH (6 to 9); however, few samples downstream from AMD sources met criteria for acidity less than alkalinity (net alkalinity = 20 mg/L as $CaCO_3$) and concentrations of dissolved iron (0.3 mg/L) and total manganese (1.0 mg/L). Iron, aluminum, and various trace elements including cobalt, copper, lead, nickel, and zinc, were present in many streamwater samples at concentrations at which continuous exposure can not be tolerated by aquatic organisms without an unacceptable effect. Furthermore, concentrations of sulfate, iron, manganese, aluminum, and (or) beryllium in some samples exceeded drinking-water standards. Other trace elements, including antimony, arsenic, barium, cadmium, chromium, selenium, silver, and thallium did not exceed water-quality criteria for protection of aquatic organisms or human health. Nevertheless, when considered together, concentrations of iron, manganese, arsenic, cadmium, chromium, copper, lead, nickel, and zinc in a majority of the streambed sediment samples from Mahanoy Creek and AMD-affected tributaries exceeded the probable effect level for toxicity because of multiple contaminants.

The concentrations of dissolved metals provided a consistent basis for evaluation of acidity and contaminant loading and were similar to concentrations of total metals in AMD water samples. With two exceptions, the computed sum of metals loading from AMD sources upstream of stream-monitoring sites generally exceeded the measured load at the stream site. Metals loading measured on Mahanoy Creek at Ashland (S16) and at the mouth of Shenandoah Creek (S14) were greater than the sum of metals from AMD sources. This result indicates that one or more AMD sources was not sampled in that reach. Additional investigation in the reaches between S13 and S16 is warranted to identify and characterize these sources. Elsewhere, smaller measured loads compared to the computed sum of metals loading from AMD sources upstream indicates that metals accumulate in upstream segments during base-flow conditions. This accumulation of metals in the stream degrades the aquatic habitat and indicates that a greater quantity of metal would need to be removed at the AMD source to achieve similar load reduction at a downstream location.

The water-quality data for the sampled AMD sources were used to determine priority ranks of the sources on the basis of loadings of iron, manganese, and aluminum and to identify possible remedial alternatives, including passive-treatment options. The ranking sequence for the top AMD sources based on the high base-flow data generally matched that based on the low base-flow data. Although concentrations increased with decreased flow, the contaminant loadings generally increased with flow; six previously identified intermittent AMD sources were not discharging during the low base-flow sampling period. The top 4 AMD sources, Locust Gap Tunnel (M29), Packer #5 Breach (M13), Packer #5 Borehole (M12), and Girard Mine seepage (M11), on the basis of dissolved metals loading in March 2001 accounted for more than 50 percent of the metals loading to Mahanoy Creek, whereas the top 15 AMD sources accounted for more than 99 percent of the metals loading. When sampled in March 2001, the top 15 AMD sources had flow rates ranging from 0.4 to 17.2 ft^3/s (680 to 29,200 L/min) and pH from 3.9 to 6.7. Dissolved iron was the principal source of acidity and metals loading; concentrations of iron ranged from 2.1 to 18 mg/L. Dissolved manganese ranged from 0.95 to 6.4 mg/L. Dissolved aluminum exceeded 1.0 mg/L at 4 of the top 15 AMD sources but was less than 0.4 mg/L at the others. Nine of the top 15 AMD sources, including the top 4, were net alkaline (alkalinity greater than acidity); the others were net acidic and will require additional alkalinity to facilitate metals removal and maintain near-neutral pH.

Alkalinity can be acquired by the dissolution of limestone and (or) bacterial sulfate reduction within various passive-treatment systems including anoxic or oxic limestone drains, limestone-lined channels, or compost wetlands. Subsequently, the gradual oxidation and consequent precipitation of iron and manganese can be accommodated within settling ponds or aerobic wetlands. Assuming an iron removal rate of 180 lb/acre/d (20 $g/m^2/d$), constructed treatment wetlands at the top 15 AMD sites would require a minimum area ranging from 0.3 to 5.8 acres (1,210 to 23,500 m^2). Land area below the Packer #5 Breach (M13; ranked 2nd), the Packer #5 Borehole (M12; ranked 3rd), and the Centralia Tunnel (M19; ranked 6) may be sufficient for installation of passive treatment. However, because of the proximity of the Locust Gap Tunnel and many other discharges to streams, roads, or railroads, and the limited availability or access to land at the discharge location, passive treatment would not be suitable at many AMD sites. The reduction of infiltration and removal of culm waste and (or) the relocation of the discharge to nearby areas could decrease the AMD quantities and facilitate treatment at some of the priority AMD sites.

Acknowledgments

Suzanne J. Ward, Jeffrey J. Chaplin, K. Samuel Fishel, and Mark A. Roland of the USGS, Martin A. Friday of PaDEP, Ryan J. Koch of Schuylkill Conservation District, and Stephen E. Ulceski of Ashland Borough assisted with data collection for the assessment. Robert Seal, Jane M. Hammarstrom, and Nadine Piatak provided chemical and mineralogical analyses of the water samples and streambed sediments. Michael D. Bilger and Robin A. Brightbill of the USGS and Martin A. Friday of PaDEP conducted the aquatic ecological surveys. Kevin J. Breen, Ralph J. Haefner, Jane M. Hammarstrom, Kim L. Otto, and J. Kent Crawford, of USGS provided helpful reviews of early drafts of this report.

References Cited

Barbour, M.T., Gerritsen, J., Snyder, B.D., and Stribling, J.B., 1999, Rapid bioassessment protocols for use in streams and wadeable rivers--periphyton, benthic macroinvertebrates and fish (2d ed.): U.S. Environmental Protection Agency EPA 841-B-99-002, 11 chapters, 4 appendixes.

Becher, A.E., 1991, Groundwater resources in and near the anthracite basins of Schuylkill and adjacent counties, Pennsylvania: Pennsylvania Geological Survey, 4th Series, Water Resources Report 64, 59 p., 1 plate.

Berg, T.M., Barnes, J.H., Seven, W.D., Skema, V.K., Wilshusen, J.P., and Yannicci, D.S., 1989, Physiographic provinces of Pennsylvania: Pennsylvania Geological Survey, 4th Series, Map 13, scale 1:2,000,000.

Berg, T.M., and others, 1980, Geologic map of Pennsylvania: Pennsylvania Geological Survey, 4th Series, Map 1, scale 1:2,500,000, 3 sheets.

Bilger, M.D., Brightbill, R.A., and Campbell, H.L., 1999, Occurrence of organochlorine compounds in whole fish tissue from streams of the Lower Susquehanna River Basin, Pennsylvania and Maryland, 1992: U.S. Geological Survey Water-Resources Investigations Report 99-4065, 17 p.

Blowes, D.W., and Ptacek, C.J., 1994, Acid-neutralization mechanisms in inactive mine tailings, in Jambor, J.L, and Blowes, D.W., eds., Environmental geochemistry of sulfide mine-wastes: Mineralogical Association of Canada, Short Course Handbook, v. 22, p. 271-292.

Breen, K.J., and Gavin, A.J., 1995, Reconnaissance of trace-metal loadings in Mahanoy Creek - - a tributary to the Lower Susquehanna River, Pennsylvania, affected by acidic mine drainage (abs.): EOS, Transactions of the American Geophysical Union, v. 76, no. 17, p. S133 (*http://wwwpah2o.er.usgs.gov/projects/lsus/abs.mahanoy.html*).

Burrows, W.D., 1977, Aquatic aluminum--chemistry, toxicology, and environmental prevalence: CRC Critical Reviews in Environmental Controls, v. 7, p. 167-216.

Burton, T.M., and Allan, J.W., 1986, Influence of pH, aluminum, and organic matter on stream invertebrates: Canadian Journal of Fisheries and Aquatic Sciences, v. 43, p. 1285-1289.

Butler, R.L., Cooper, E.L., Crawford, J.K., Hales, D.C., Kimmel, W.G., and Wagner, C.C., 1973, Fish and food organisms in acid mine waters of Pennsylvania: U.S. Environmental Protection Agency EPA-R3-73-032, 158 p.

Campbell, P.G.C., Lewis, A.G., Chapman, P.M., Crowder, A.A., Fletcher, W.K., Imber, B., Luoma, S.N., Stokes, P.M., and Winfrey, M.,1988, Biologically available metals in sediments: National Research Council Canada Publication No. NRCC 27694, 298 p.

Cherry, D.S., Currie, R.J., Soucek, D.J., Latimer, H.A., and Trent, G.C., 2001, An integrative assessment of a watershed impacted by abandoned mined land discharges: Environmental Pollution, v. 111, p. 377-388.

Commonwealth of Pennsylvania, 1998a, Chapter 87. Surface mining of coal. Pennsylvania Code, Title 25. Environmental Protection: Harrisburg, Pa., Commonwealth of Pennsylvania, p. 87.1-87.122.

Commonwealth of Pennsylvania, 1998b, Chapter 89. Underground mining of coal and coal preparation facilities. Pennsylvania Code, Title 25. Environmental Protection: Harrisburg, Pa., Commonwealth of Pennsylvania, p. 89.1-89.96.

Commonwealth of Pennsylvania, 2001a, Chapter 96. Water quality standards implementation. Pennsylvania Code, Title 25. Environmental Protection: Harrisburg, Pa., Commonwealth of Pennsylvania, p. 96.1-96.8.

Commonwealth of Pennsylvania, 2001b, Chapter 16. Water Quality Toxics Management Strategy--Statement of policy. Pennsylvania Code, Title 25. Environmental Protection: Harrisburg, Pa., Commonwealth of Pennsylvania, p. 16.1-16.46.

Commonwealth of Pennsylvania, 2002, Chapter 93. Water Quality Standards. Pennsylvania Code, Title 25. Environmental Protection: Harrisburg, Pa., Commonwealth of Pennsylvania, p. 93.1-93.226.

Coston, J.A., Fuller, C.C., and Davis, J.A., 1995, Pb^{2+} and Zn^{2+} adsorption by a natural aluminum- and iron-bearing surface coating on an aquifer sand: Geochimica et Cosmochimica Acta, v. 59, p. 3535-3547.

Cravotta, C.A. III, 1991, Geochemical evolution of acidic ground water at a reclaimed surface coal mine in western Pennsylvania, in Proceedings of the 1991 National Meeting of the American Society of Surface Mining and Reclamation, May 14-17, 1991, Durango, Colorado: American Society of Mining and Reclamation, p. 43-68.

Cravotta, C.A. III, 2002, New method to estimate longevity of anoxic limestone drains (poster), in Proceedings of the 19th Annual Meeting of American Society of Mining and Reclamation, Lexington, Kentucky, June 9-13, 2002: American Society of Mining and Reclamation, p. 1062-1064.

Cravotta, C.A. III, 2003, Size and performance of anoxic limestone drains to neutralize acidic mine drainage: Journal of Environmental Quality, v. 32, p. 1277-1289.

Cravotta, C.A. III, and Bilger, M.D., 2001, Water-quality trends for a stream draining the Southern Anthracite Field, Pennsylvania: Geochemistry-Exploration, Environment, Analysis, v. 1, p. 33-50.

Cravotta, C.A. III, Brady, K.B.C., Rose, A.W., and Douds, J.B., 1999, Frequency distribution of the pH of coal-mine drainage in Pennsylvania, in Morganwalp, D.W., and Buxton, H., eds., U.S. Geological Survey Toxic Substances Hydrology Program--Proceedings of the technical meeting: U.S. Geological Survey Water-Resources Investigations Report 99-4018A, p. 313-324.

Cravotta, C.A. III, Breen, K.J., and Seal, R., 2001, Arsenic is ubiquitous but not elevated in abandoned coal-mine discharges in Pennsylvania (abs.), in U.S. Geological Survey Appalachian Region Integrated Science Workshop Proceedings, Gatlinburg, Tennessee, October 22-26, 2001: U.S. Geological Survey Open-File Report 01-406, p. 105.

Cravotta, C.A. III, and Kirby, C.S., 2004a, Effects of abandoned coal-mine drainage on streamflow and water quality in the Schuylkill Creek Basin, Northumberland and Columbia Counties, Pennsylvania, 1999-2001: U.S. Geological Survey Water-Resources Investigations Report 03-4311, 53 p.

Cravotta, C.A. III, and Kirby, C.S., 2004b, Acidity and alkalinity in mine drainage--practical considerations, in 2004 National Meeting of the American Society of Mining and Reclamation and the 25th West Virginia Surface Mine Drainage Task Force, April 18-24, 2004: American Society of Mining and Reclamation, p. 334-365.

Cravotta, C.A. III, and Trahan, M.K., 1999, Limestone drains to increase pH and remove dissolved metals from acidic mine drainage: Applied Geochemistry, v. 14, no. 5, p. 581-606.

Cravotta, C.A. III, Ward, S.J., Koury, D.J., and Koch, R.D., 2004, Optimization of limestone drains for long-term treatment of acidic mine drainage, Swatara Creek Basin, Schuylkill County, PA in 2004 National Meeting of the American Society of Mining and Reclamation and the 25th West Virginia Surface Mine Drainage Task Force, April 18-24, 2004: American Society of Mining and Reclamation, p. 366-411.

Crock, J.G., Arbogast, B.F., and Lamothe, P.J., 1999, Laboratory methods for the analysis of environmental samples, in Plumlee G.S., and Logsdon, M.J., eds., The Environmental Geochemistry of Mineral Deposits--Part A. Processes, techniques, and health issues: Society of Economic Geologists, Reviews in Economic Geology, v. 6A, p. 265-287.

Dempsey, B.A., Paksuchon, B., Suhataikul, R., and Dietz, J.M., 2002, Sampling strategies for TMDL of AMD-affected streams, in Proceedings of the 19th Annual Meeting of American Society of Mining & Reclamation, Lexington, Kentucky, June 9-13, 2002: American Society of Mining & Reclamation, p. 10-26.

Dietz, J.M., and Dempsey, B.A., 2002, Innovative treatment of alkaline mine drainage using recirculated iron oxides in a complete mix reactor, in Proceedings of the 19th Annual Meeting of American Society of Mining & Reclamation, Lexington, Kentucky, June 9-13, 2002: American Society of Mining & Reclamation, p. 496-516.

Durlin, R.R., and Schaffstall, W.P., 1994, Water resources data, Pennsylvania, water year 1993, volume 2. Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Data Report PA-93-2, p. 271-273.

Earle, J., and Callaghan, T., 1998. Effects of mine drainage on aquatic life, water uses, and man- made structures, in Brady, K.B.C., Smith, M.W., and Schueck, J.H., eds., Coal Mine Drainage Prediction and Pollution Prevention in Pennsylvania: Harrisburg, Pa., Pennsylvania Department of Environmental Protection, 5600-BK-DEP2256, p. 4.1-4.10.

Eggleston, J.R., Kehn, T.M., and Wood, G.H., Jr., 1999, Anthracite, in Schultz, C.H., ed., The geology of Pennsylvania: Pennsylvania Geological Survey, 4th series, Special Publication 1, p. 458-469.

Elder, J.F., 1988, Metal biogeochemistry in surface-water systems--a review of principles and concepts: U.S. Geological Survey Circular 1013, 43 p.

Fishman, M.J., 1993, Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory--Determination of inorganic and organic constituents in water and fluvial sediments: U.S. Geological Survey Open-File Report 93-125, 217 p.

Fishman, M.J., and Friedman, L.C., eds., 1989, Methods for determination of inorganic substances in water and fluvial sediments: U.S. Geological Survey Techniques of Water-Resources Investigations, book 5, chap. A1, 545 p.

Hedin, R.S., Nairn, R.W., and Kleinmann, R.L.P., 1994, Passive treatment of coal mine drainage: U.S. Bureau of Mines Information Circular 9389, 35 p.

Herlihy, A.T., Kaufmann, P.R., Mitch, M.E., and Brown, D.D., 1990, Regional estimates of acid mine drainage impact on streams in the mid-Atlantic and southeastern United States: Water, Air, and Soil Pollution, v. 50, p. 91-107.

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

Hoffman, G.L., 1996, Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory--Preparation procedure for aquatic biological material determined for trace metals: U.S. Geological Survey Open-File Report 96-362, 42 p.

Hyman, D.M., and Watzlaf, G.R., 1997, Metals and other components of coal mine drainage as related to aquatic life standards, in Proceedings of the 1997 National Meeting of the American Society for Surface Mining and Reclamation, May 10-15, 1997, Austin, Texas: Princeton, W.Va., American Society for Surface Mining and Reclamation, p. 531-545.

Kirby, C.S., and Cravotta, C.A., III, 2004, Acidity and alkalinity in mine drainage--theoretical considerations, in 2004 National Meeting of the American Society of Mining and Reclamation and the 25th West Virginia Surface Mine Drainage Task Force, April 18-24, 2004: American Society of Mining and Reclamation, p. 1076-1093.

Kleinmann, R.L.P., and Watzlaf, G.R., 1986, Should the discharge standards for manganese be reexamined?, in Proceedings 1986 Symposium on Mining, Hydrology, Sedimentology, and Reclamation: Lexington, Ky., University of Kentucky Office of Engineering Services, p. 173-179.

Langmuir, Donald, 1997, Aqueous environmental geochemistry: New Jersey, Prentice-Hall, 600 p.

Lowe, T.P., May, T.W., Brumbaugh, W.G., and Kane, D.A., 1985, National Contaminant Biomonitoring Program--concentration of seven elements in freshwater fish, 1978-1981: Archives of Environmental Contamination and Toxicology, v. 14, p. 363-388.

MacDonald, D.D., Ingersoll, C.G., and Berger, T.A., 2000, Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems: Archives of Environmental Contamination and Toxicology, v. 39, p. 20-31.

McKenzie, R.M. 1980, The adsorption of lead and other heavy metals on oxides of manganese and iron: Australian Journal of Soil Research, v. 18, p. 61-73.

Meador, M.R., Hupp, C.R., Cuffney, T.F., and Gurtz, M.E., 1993a, Methods for characterizing stream habitat as part of the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 93-408, 48 p.

Meador, M.R., Cuffney, T.F., and Gurtz, M.E., 1993b, Methods for sampling fish communities as part of the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 93-104, 40 p.

Nordstrom, D.K., and Alpers, C.N., 1999, Geochemistry of acid mine waters, in Plumlee, G. S., and Logsdon, M. J., eds., The Environmental Geochemistry of Mineral Deposits--Part A. Processes, methods, and health issues: Reviews in Economic Geology, v. 6A, p. 133-160.

Pennsylvania Coal Association, 2001, Pennsylvania coal data 2000: Harrisburg, Pa., Pennsylvania Coal Association, 36 p.

Pennsylvania Department of Environmental Protection, 1998, Commonwealth of Pennsylvania 1998 Water Quality Assessment: Division of Assessment and Standards: Pennsylvania Department of Environmental Protection, Bureau of Water Quality Management, April 1998, accessed December 15, 2002, on the World Wide Web at *http://www.dep.state.pa.us/dep/deputate/watermgt/WC/Subjects/WQStandards/305_wq98_narr.htm#PART3*.

Pennsylvania Department of Environmental Protection, 1999, Pennsylvania's nonpoint source management program, 1999 Update: Harrisburg, Pa., Pennsylvania Department of Environmental Protection, 394-2000-002, p. 42-51.

Pennsylvania Department of Environmental Protection, 2001, Watershed Restoration Action Strategy (WRAS) Subbasin 06B Mahanoy Creek and Shamokin Creek Watersheds (Susquehanna River) Northumberland and Schuylkill Counties, PA: Harrisburg, Pa., Pennsylvania Department of Environmental Protection, 19 p.

Pennsylvania Department of Environmental Protection, 2002, Mahanoy Creek Watershed TMDL Columbia, Northumberland and Schuylkill Counties: Pennsylvania Department of Environmental Protection, Bureau of Watershed Management, October 1, 2002, accessed May 25, 2004, on the World Wide Web at *http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/MahanoyCr_TMDL.pdf.*

Pennsylvania Department of Environmental Protection, 2004, Proposed Section 303(d) list: Pennsylvania Department of Environmental Protection, Bureau of Water Quality Management, accessed February 15, 2004, on the World Wide Web at *http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/303d-Report.htm*.

Persaud, D., Jaagumagi, R., and Hayton, A., 1993, Guidelines for the protection and management of aquatic sediment quality in Ontario: Ontario Ministry of the Environment, Water Resources Branch, 24 p.

Rantz, S.E., and others, 1982a, Measurement and computation of streamflow--1. Measurement of stage and discharge: U.S. Geological Survey Water-Supply Paper 2175, 284 p.

Rantz, S.E., and others, 1982b, Measurement and computation of streamflow--2. Computation of discharge: U.S. Geological Survey Water-Supply Paper 2175, 631 p.

Reed, L.A., Beard, M.M., Growitz, D.J., 1987, Quality of water in mines in the western middle coal field, anthracite region, east-central Pennsylvania: U.S. Geological Survey Water-Resources Investigations Report 85-4038, 51 p.

Rice, K.C., 1999, Trace-element concentrations in streambed sediment across the conterminous United States: Environmental Science & Technology, v. 33, p. 2499-2504.

Rose, A.W., and Cravotta, C.A., III, 1998, Geochemistry of coal-mine drainage, in Brady, K.B.C., Smith, M.W., and Schueck., J., eds., Coal Mine Drainage Prediction and Pollution Prevention in Pennsylvania: Harrisburg, Pa., Pennsylvania Department of Environmental Protection, 5600-BK-DEP2256, p. 1.1-1.22.

Sanders & Thomas, Inc., 1975, Operation Scarlift project no. SL-197, Mahanoy Creek mine drainage pollution abatement project: Harrisburg, Pa., Sanders & Thomas, Inc. Engineers, 125 p., 1 appendix, 2 plates.

Schueck, J.H., Helfrich, D.R., and Fromell, D.J., 2004, Limestone upflow pond with siphon discharge design considerations--a simple solution to high volume, high metals AMD discharges, in Proceedings of the 6th Annual Statewide Conference on Abandoned Mine Reclamation, June 9-12, 2004, Indiana PA: Bedford, Pa., Western Pennsylvania Coalition for Abandoned Mine Reclamation, 23 p.

Skousen, J.G., Rose, A.W., Geidel, G., Foreman, J., Evans, R., Hellier, W., and others, 1998, Handbook of technologies for avoidance and remediation of acid mine drainage: Morgantown, W.Va., National Mine Land Reclamation Center, 131 p.

Taggart, J.E., Jr., 2002, Analytical methos for chemical analysis of geologic and other materials, U.S. Geological Survey: U.S. Geological Survey Open-File Report 02-0223 (*http://pubs.usgs.gov/of/2002/ofr-02-0223/*).

Tonkin, J.W., Balistrieri, L.S., and Murray, J.W., 2004, Modeling sorption of divalent metal cations on hydrous manganese oxide using the diffuse double layer model: Applied Geochemistry, v. 19, p. 29-53.

U.S. Environmental Protection Agency, 1993, Fish field and laboratory methods for evaluating the biological integrity of surface waters: U.S. Environmental Protection Agency EPA 600/R-92/111, 348 p.

U.S. Environmental Protection Agency, 1997, Guidance for assessing chemical contaminant data for use in fish advisories--Volume 2, Risk assessment and fish consumption limits (2d ed.): Philadelphia, U.S. Environmental Protection Agency EPA 823-B-97-009, accessed December 15, 2002, on the World Wide Web at *http://www.epa.gov/OST/fish/volume2.htm*.

U.S. Environmental Protection Agency, 2002a, National recommended water quality criteria--2002: U.S. Environmental Protection Agency EPA 822-R-02-047, 33 p.

U.S. Environmental Protection Agency, 2002b, National primary drinking water standards: U.S. Environmental Protection Agency EPA 816-F-02-013, July 2002, 7 p., accessed December 15, 2002, on the World Wide Web at *http://www.epa.gov/safewater*.

U.S. Geological Survey, 1997 to present, National field manual for the collection of water-quality data: U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chaps. A1-A9, 2 vols., variously paged, accessed December 15, 2002, on the World Wide Web at *http://pubs.water.usgs.gov/twri9A*.

U.S. Geological Survey, 2000, Land cover: U.S. Geological Survey Map 66, scale 1:500,000, accessed December 15, 2002, on the World Wide Web at *http://edcwww.cr.usgs.gov/programs/lccp/nationallandcover.html*.

U.S. Office of Surface Mining Reclamation and Enforcement, 2002, AMDTreat 3.1: Washington, D.C., U.S. Office of Surface Mining Reclamation and Enforcement, accessed June 15, 2003, on the World Wide Web at *http://amd.osmre.gov*.

Vinci, B.J., and Schmidt, T.W., 2001, Passive, periodic flushing technology for mine drainage treatment systems, in Proceedings of the 18th Annual Meeting of American Society for Surface Mining and Reclamation, Albuquerque, New Mexico, June 3-7, 2001: American Society for Surface Mining and Reclamation, p. 611-625.

Watzlaf, G.R., Schroeder, K.T., and Kairies, C., 2000, Long-term performance of alkalinity- producing passive systems for the treatment of mine drainage, in Proceedings of the 2000 National Meeting of the American Society for Surface Mining and Reclamation: Princeton, W.Va., American Society for Surface Mining and Reclamation, p. 262-274.

Way, J.H., 1999, Appalachian Mountain section of the Ridge and Valley province, in Schultz, C.H., ed., The geology of Pennsylvania: Pennsylvania Geological Survey, 4th series, Special Publication 1, p. 352-361.

Weaver, K.R., Lagnese, K.M., and Hedin, R.S., 2004, Technology and design advances in passive treatment system flushing, in 2004 National Meeting of the American Society of Mining and Reclamation and the 25th West Virginia Surface Mine Drainage Task Force, April 18-24, 2004: American Society of Mining and Reclamation, p. 1974-1989.

Webster, J.G., Swedlund, P.J., and Webster, K.S., 1998, Trace metal adsorption onto an acid mine drainage iron(III) oxy hydroxy sulfate: Environmental Science & Technology, v. 32, no. 10, p. 1361-1368.

Williams, D.R., Sams, J.I., III, and Mulkerrin, M.E., 1996, Effects of coal-mine discharges on the quality of Stonycreek River and its tributaries, Somerset and Cambria Counties, Pennsylvania: U.S. Geological Survey Water-Resources Investigations Report 96-4133, 95 p.

Williams, D.R., Clark, M.E., and Brown, J.B., 1999, stream water quality in coal-mined areas of the Lower Cheat River Basin, West Virginia and Pennsylvania, during low-flow conditions, July 1997: U.S. Geological Survey Water-Resources Investigations Report 98-4258, 8 p.

Winland, R.L., Traina, S.J., and Bigham, J.M., 1991, Chemical composition of ochreous precipitates from Ohio coal mine drainage: Journal of Environmental Quality, v. 20, p. 452-460.

Winterbourn, J.J., McDiffett, W.R., and Eppley, S.J., 2000, Aluminium and iron burdens of aquatic biota in New Zealand streams contaminated by acid mine drainage--effects of trophic level: The Science of the Total Environment, v. 254, p. 45-54.

Wood, C.R., 1996, Water quality of large discharges in the anthracite region of eastern Pennsylvania: U.S. Geological Survey Water-Resources Investigations Report 95-4243, 68 p.

Wood, G.H., Jr., Kehn, T.M., and Eggleston, J.R., 1986, Deposition and structural history of the Pennsylvania Anthracite region, in Lyons P.C. and Rice C.L., eds., Paleoenvironmental and tectonic controls in coal-forming basins of the United States: Geological Society of America Special Paper 210, p. 31-47.

Wood, W.W., 1976, Guidelines for the collection and field analysis of ground-water samples for selected unstable constituents: U.S. Geological Survey Techniques of Water-Resources Investigations, book 1, chap. D2, 24 p.

Appendixes

Appendix A.

Hydrological data for abandoned mine drainage and associated stream monitoring sites in the Mahanoy Creek Basin, March, August, and October 2001 (CD-ROM, only)

Appendix B.
Geographic Information System (GIS) project with linked water-quality data base, Mahanoy Creek Basin, March, August, and October 2001 (CD-ROM, only)

Appendix C.
Digital reproductions of out-of-print reports (CD-ROM, only)

Sanders & Thomas, Inc., 1975, Operation Scarlift project no. SL-197, Mahanoy Creek mine drainage pollution abatement project: Harrisburg, Pa., Sanders & Thomas, Inc. Engineers, 125 p., 1 appendix, 2 plates (compiled from digital documents available at *http://www/amrclearinghouse.org/Sub/SCARLIFTReports/Mahanoy*).

Growitz, D.J., Reed, L.A., and Beard, M.M., 1985, Reconnaissance of mine drainage in the coal fields of eastern Pennsylvania: U.S. Geological Survey Water-Resource Investigations Report 83-4274, 54 p.

Reed, L.A., Beard, M.M., Growitz, D.J., 1987, Quality of water in mines in the western middle coal field, anthracite region, east-central Pennsylvania: U.S. Geological Survey Water-Resources Investigations Report 85-4038, 51 p.

Wood, C.R., 1996, Water quality of large discharges in the anthracite region of eastern Pennsylvania, U.S. Geological Survey Water-Resources Investigations Report 95-4243, 68 p.

Appendix D.
"AMD Treat" computation worksheets for Centralia Mine Tunnel Discharge (M19), Mahanoy Creek Basin, March and August 2001 (CD-ROM, only)