

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

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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.	
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-Resources Investigations Report 83-4274, 54 p.	
Reed, L.A., Beard, M.M., and 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.	
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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 ($^{\circ}\text{C}$) may be converted to degrees Fahrenheit ($^{\circ}\text{F}$) as follows:

$$^{\circ}\text{F} = (1.8 \times ^{\circ}\text{C}) + 32$$

Temperature in degrees Fahrenheit ($^{\circ}\text{F}$) may be converted to degrees Celsius ($^{\circ}\text{C}$) as follows:

$$^{\circ}\text{C} = (^{\circ}\text{F} - 32) / 1.8$$

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 ($\mu\text{g}/\text{L}$). Concentrations of constituents in sediment are given as micrograms per gram ($\mu\text{g}/\text{g}$) or weight percent (wt%).

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius ($\mu\text{S}/\text{cm}$ at 25°C).

Symbols are used to indicate less than (<), less than or equal to (\leq), greater than (>), greater than or equal to (\geq), equal to (=), and approximately equal to (\sim).

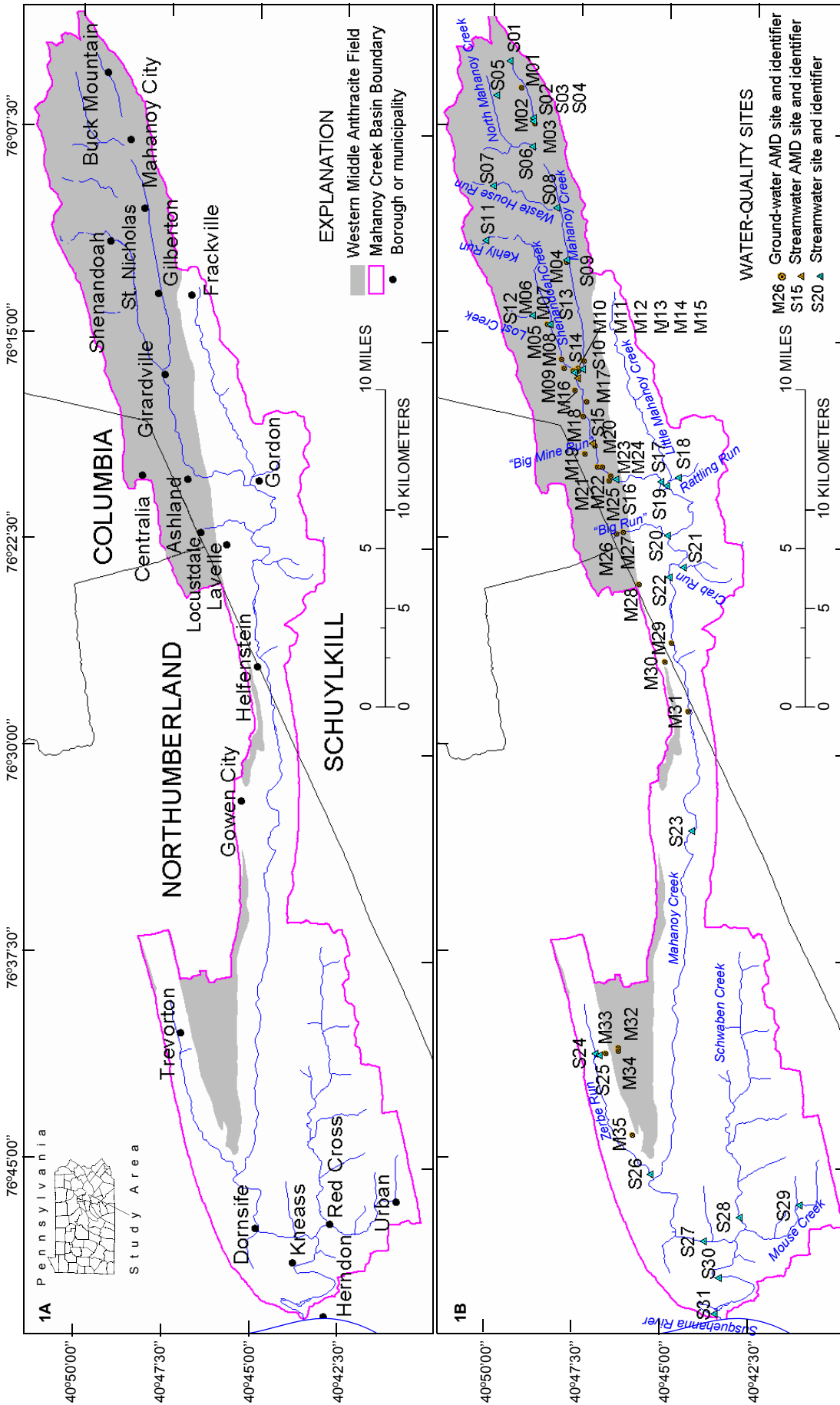


Figure 1. Mahanoy Creek Basin, Pennsylvania, showing locations of (A) municipalities and Western Middle Anthracite Field, (B) named streams and corresponding streamwater and mine-drainage monitoring locations. Mine-drainage sites that discharge directly from ground designated ground-water AMD, and those 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.

face 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).

mated 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) inconsis-

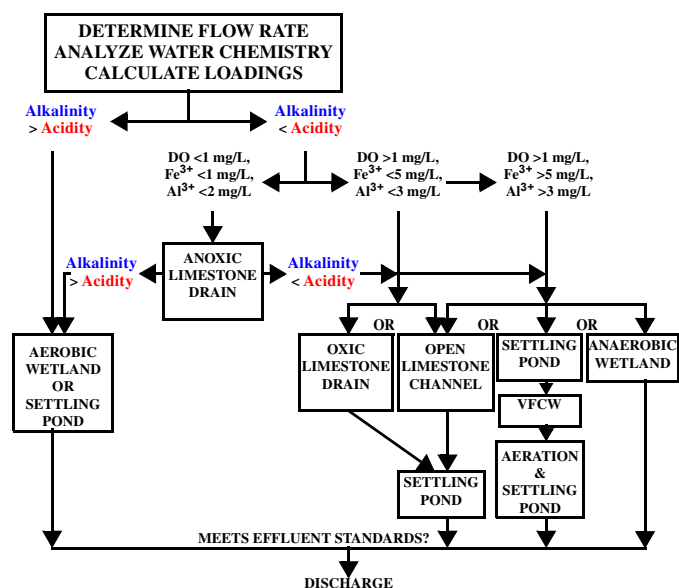


Figure 2. Flow chart for selection of passive-treatment alternatives modified from Hedin and others (1994), Skousen and others (1998), and Pennsylvania Department of Environmental Protection (1999). Vertical flow compost wetland (VFCW) also known as SAPS or RAPS. Less than (<), greater than (>), milligrams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

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tency 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-m² 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; 40°49'16.6" represents 40°49'16.6" north latitude and 76°06'17.4" represents 76°06'17.4" west longitude]

Site name and local identification number ¹	Published site identification number			Latitude	Longitude	Water-shed 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 Douyville 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}	Abandoned mine drainage sources ³	Scarflift 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. Scarflift 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://www.pah2o.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

$$\text{Metal load} = fQ(C_{\text{Fe}} + C_{\text{Mn}} + C_{\text{Al}}), \quad (6)$$

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

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

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

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

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 remaining 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/m²/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.

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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 ft³/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 ft³/s (3,380 to 15,600 L/min), respectively. Flow rates of AMD ranged from 0 to 17.2 ft³/s (0 to 29,200 L/min); the medians for the low and high base-flow surveys were 0.09 and 0.53 ft³/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),

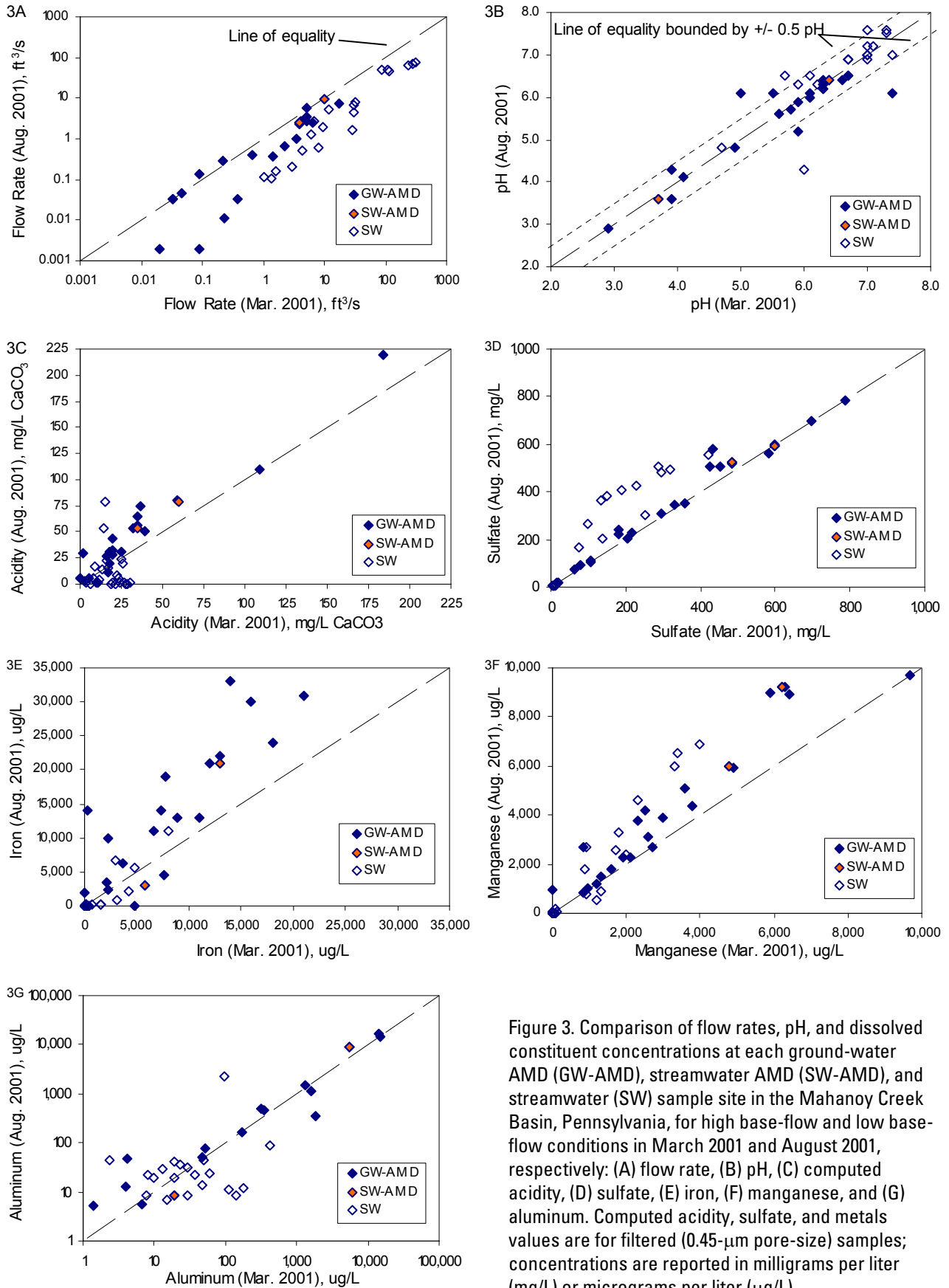


Figure 3. Comparison of flow rates, pH, and dissolved constituent concentrations at each ground-water AMD (GW-AMD), streamwater AMD (SW-AMD), and streamwater (SW) sample site in the Mahanoy Creek Basin, Pennsylvania, for high base-flow and low base-flow conditions in March 2001 and August 2001, respectively: (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 ($\mu\text{g/L}$).

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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 $\mu\text{g/L}$) 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

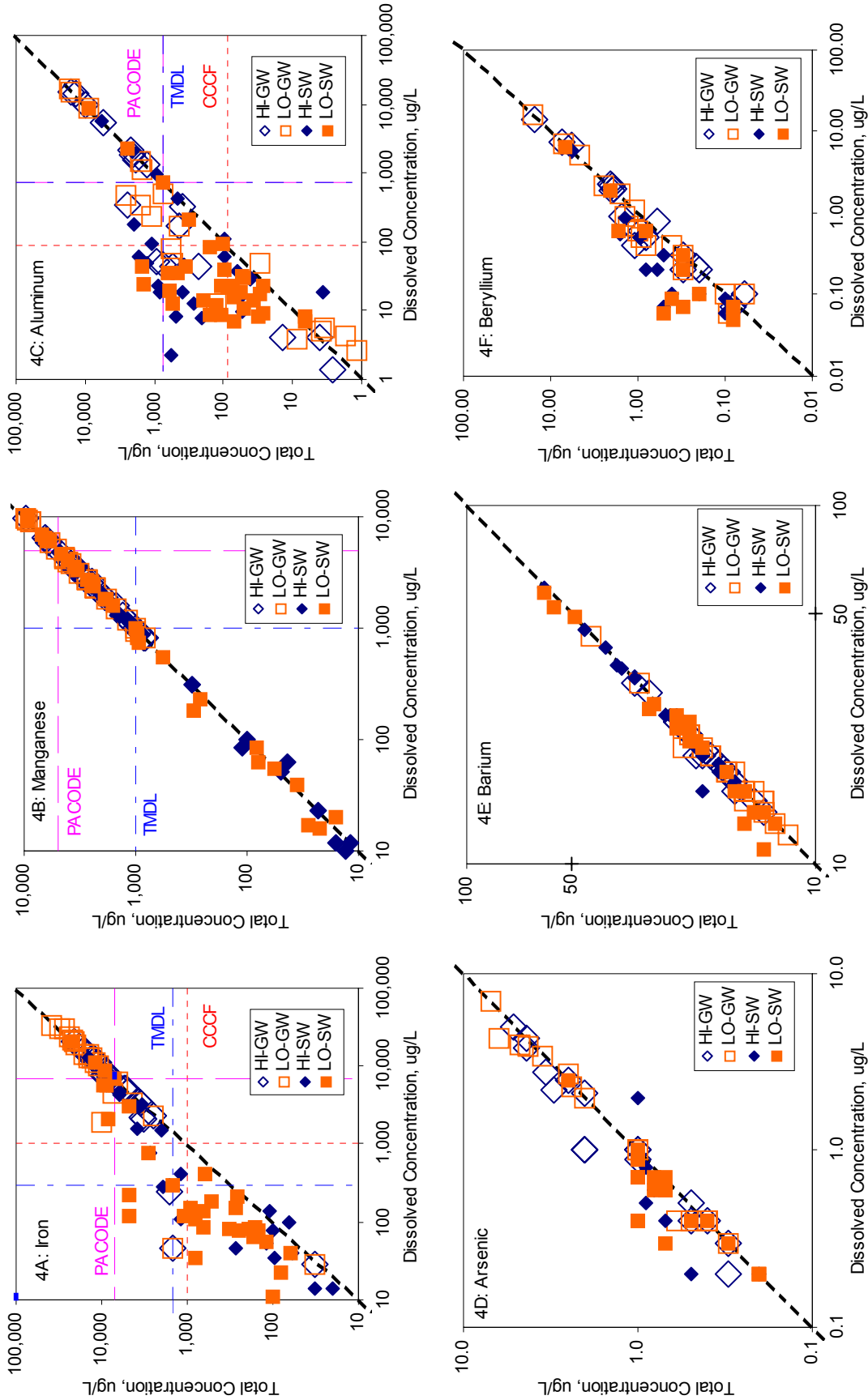


Figure 4. Comparison of total (whole-water) and dissolved (0.45- μ m pore-size filter) concentrations, in micrograms per liter (μ g/L), for low base-flow and high base-flow abandoned-mine drainage (HI-GW; LO-GW) and streamwater (HI-SW; LO-SW) samples collected in Mahanoy Creek Basin, Pennsylvania, March 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).

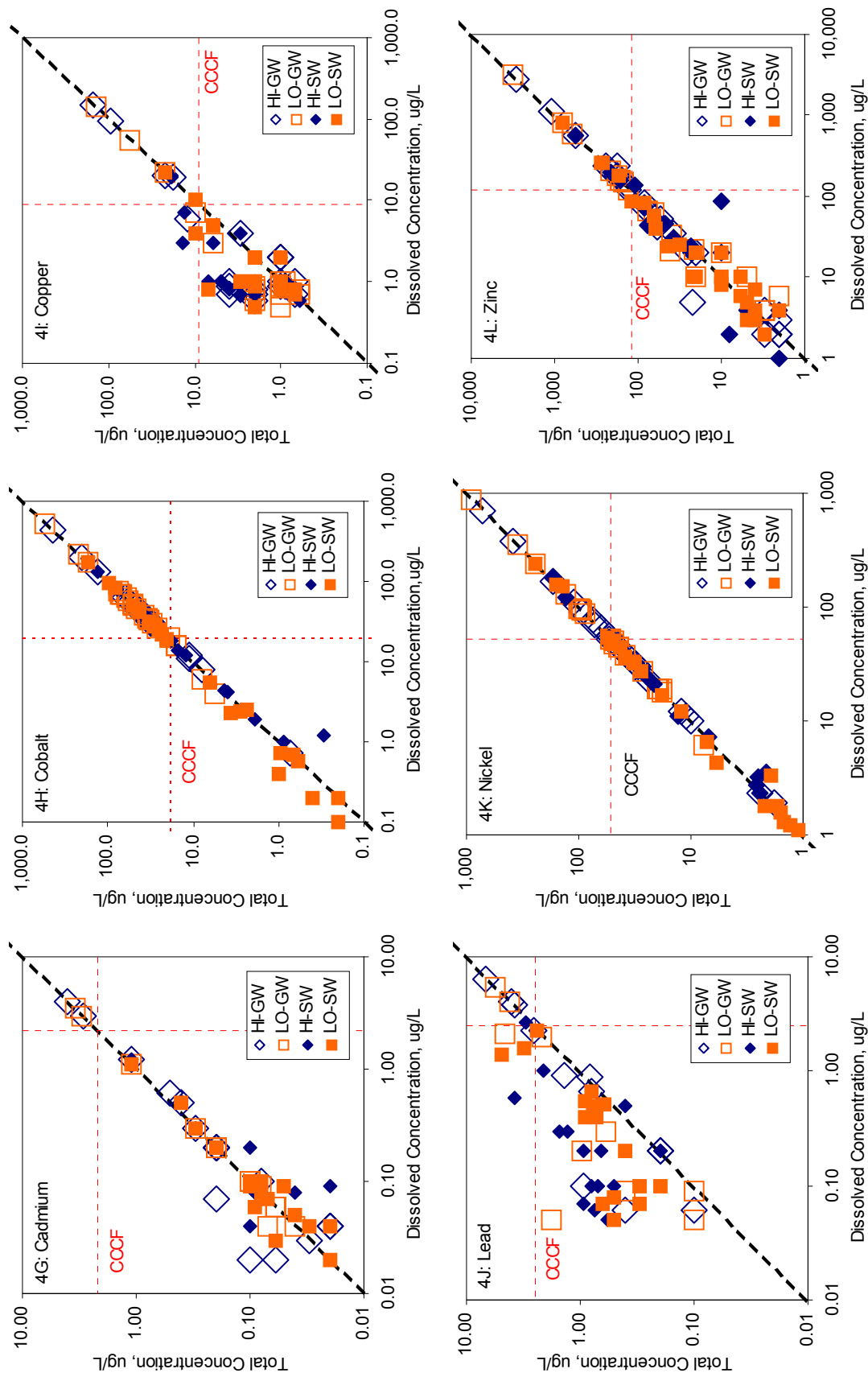


Figure 4. Comparison of total (whole-water) and dissolved (0.45- μ m pore-size filter) concentrations, in micrograms per liter (μ g/L), for low base-flow and high base-flow abandoned-mine drainage (HI-GW; LO-GW) and streamwater (HI-SW; LO-SW) samples collected in Mahanoy Creek Basin, Pennsylvania, March 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 stream-

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water 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 CaCO₃) 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 ft³/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 ochrestained channel of the sampled reach for S04 was flowing at 8.9 ft³/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 ft³/s (14,800 L/min) with pH

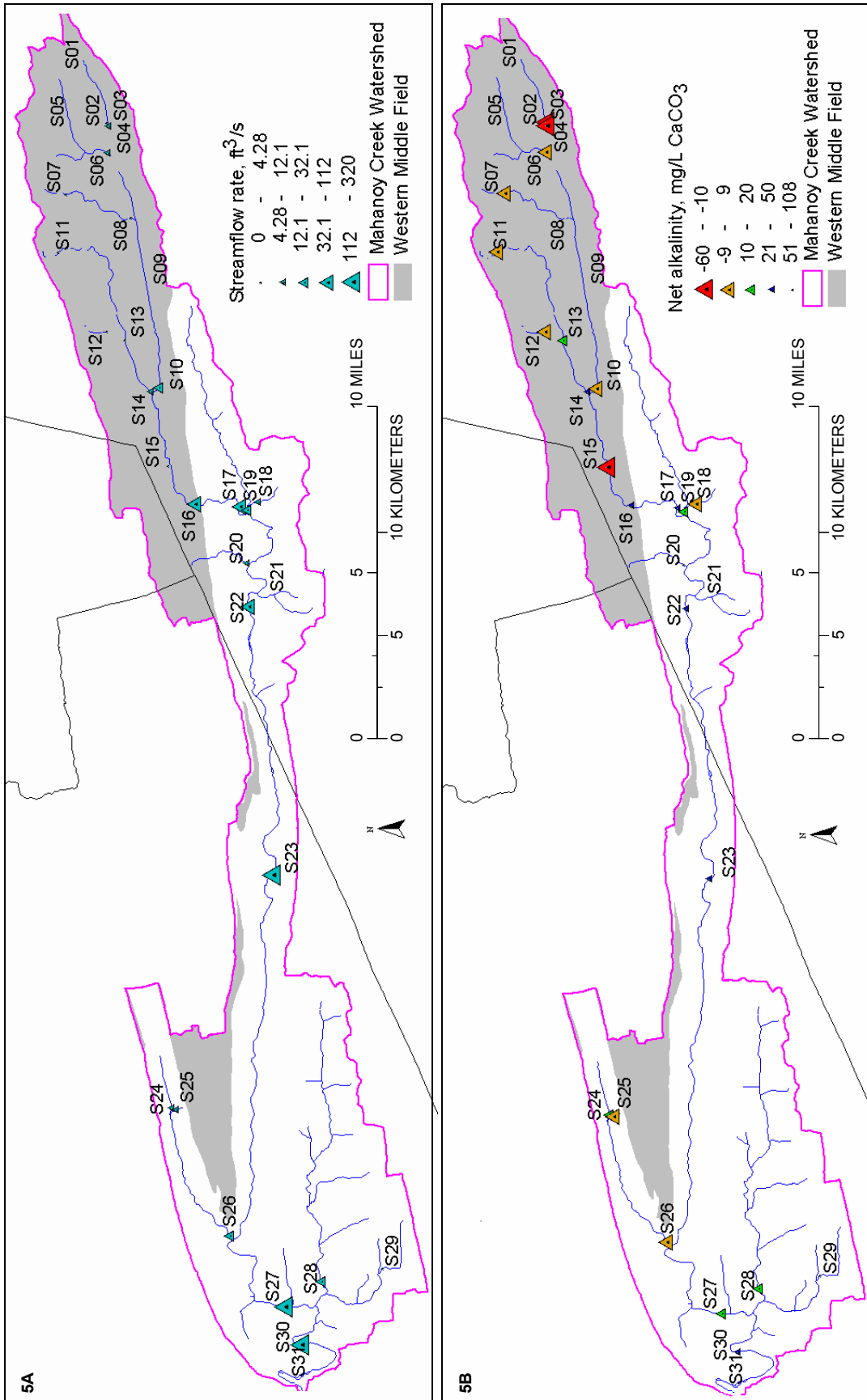


Figure 5. Mahanoy Creek Basin, Pennsylvania, showing: (A) streamflow; concentrations of (B) net alkalinity, (C) dissolved iron, (D) dissolved manganese, (E) dissolved aluminum, (F) sulfate, and (G) nitrate; and (H) Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity at streamwater sampling 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 (ft³/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L CaCO₃); and dissolved metals, sulfate, and nitrate concentrations in milligrams per liter (mg/L).

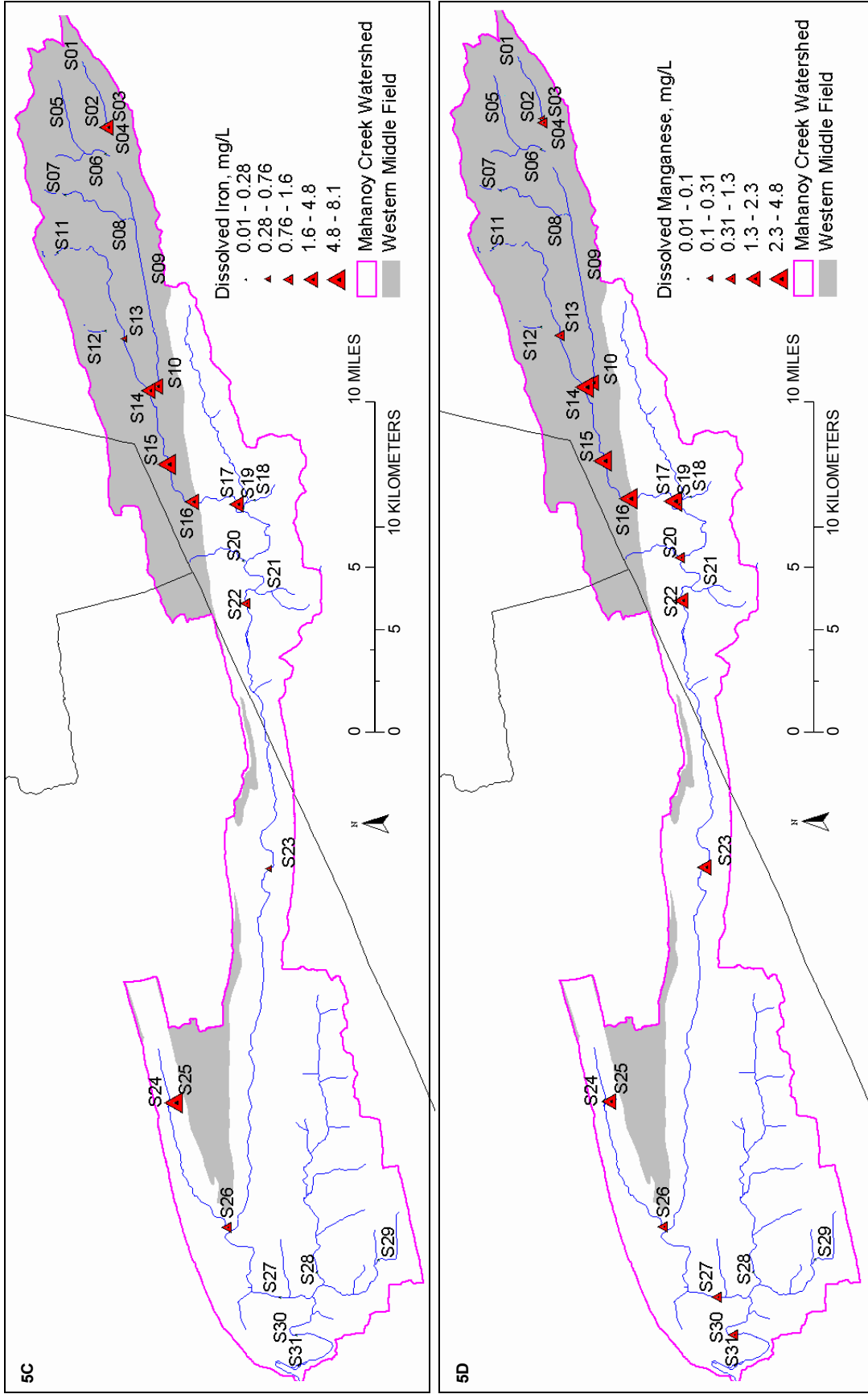


Figure 5. Mahanoy Creek Basin, Pennsylvania, showing: (A) streamflow; concentrations of (B) net alkalinity, (C) dissolved iron, (D) dissolved manganese, (E) dissolved aluminum, (F) sulfate, and (G) nitrate; and (H) Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity at streamwater sampling 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 (ft³/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L CaCO₃); and dissolved metals, sulfate, and nitrate concentrations in milligrams per liter (mg/L).—Continued

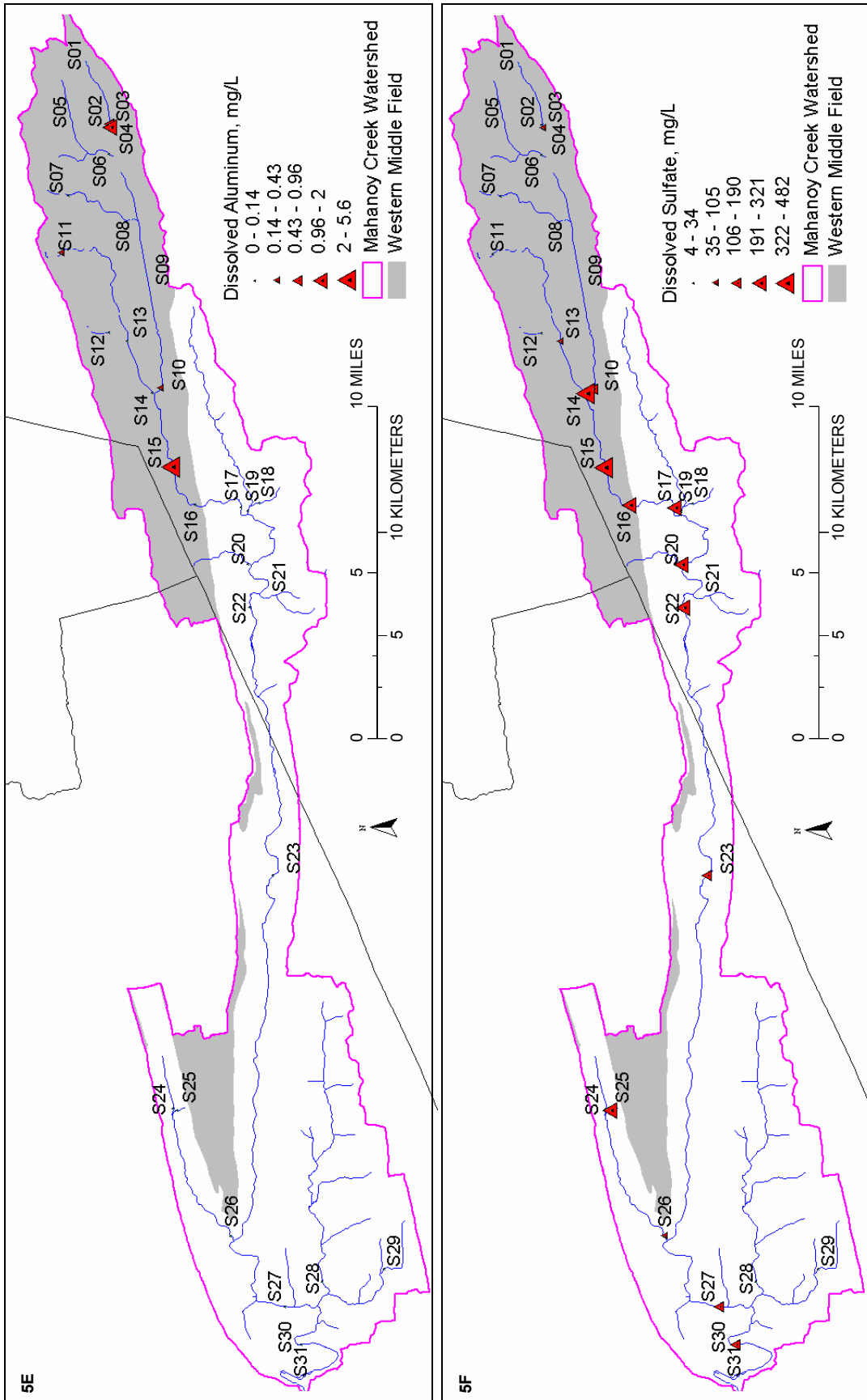


Figure 5. Mahanoy Creek Basin, Pennsylvania, showing: (A) streamflow; concentrations of (B) net alkalinity, (C) dissolved iron, (D) dissolved manganese, (E) dissolved aluminum, (F) sulfate, and (G) nitrate; and (H) Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity at streamwater sampling 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 (ft³/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L CaCO₃); and dissolved metals, sulfate, and nitrate concentrations in milligrams per liter (mg/L).—Continued

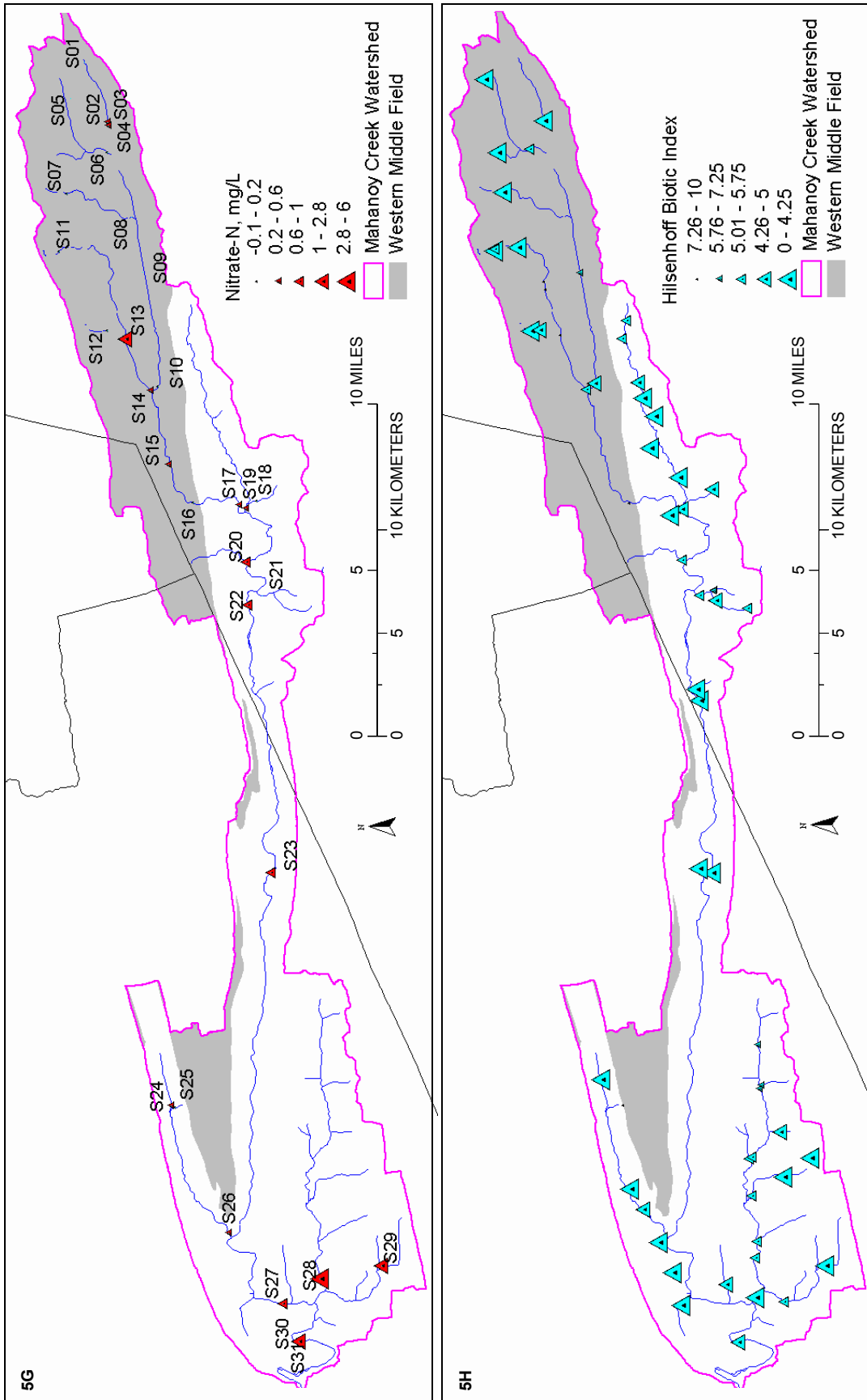


Figure 5. Mahanoy Creek Basin, Pennsylvania, showing: (A) streamflow; concentrations of (B) net alkalinity, (C) dissolved iron, (D) dissolved manganese, (E) dissolved aluminum, (F) sulfate, and (G) nitrate; and (H) Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity at streamwater sampling 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 (ft³/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L CaCO₃); and dissolved metals, sulfate, and nitrate concentrations in milligrams per liter (mg/L).—Continued

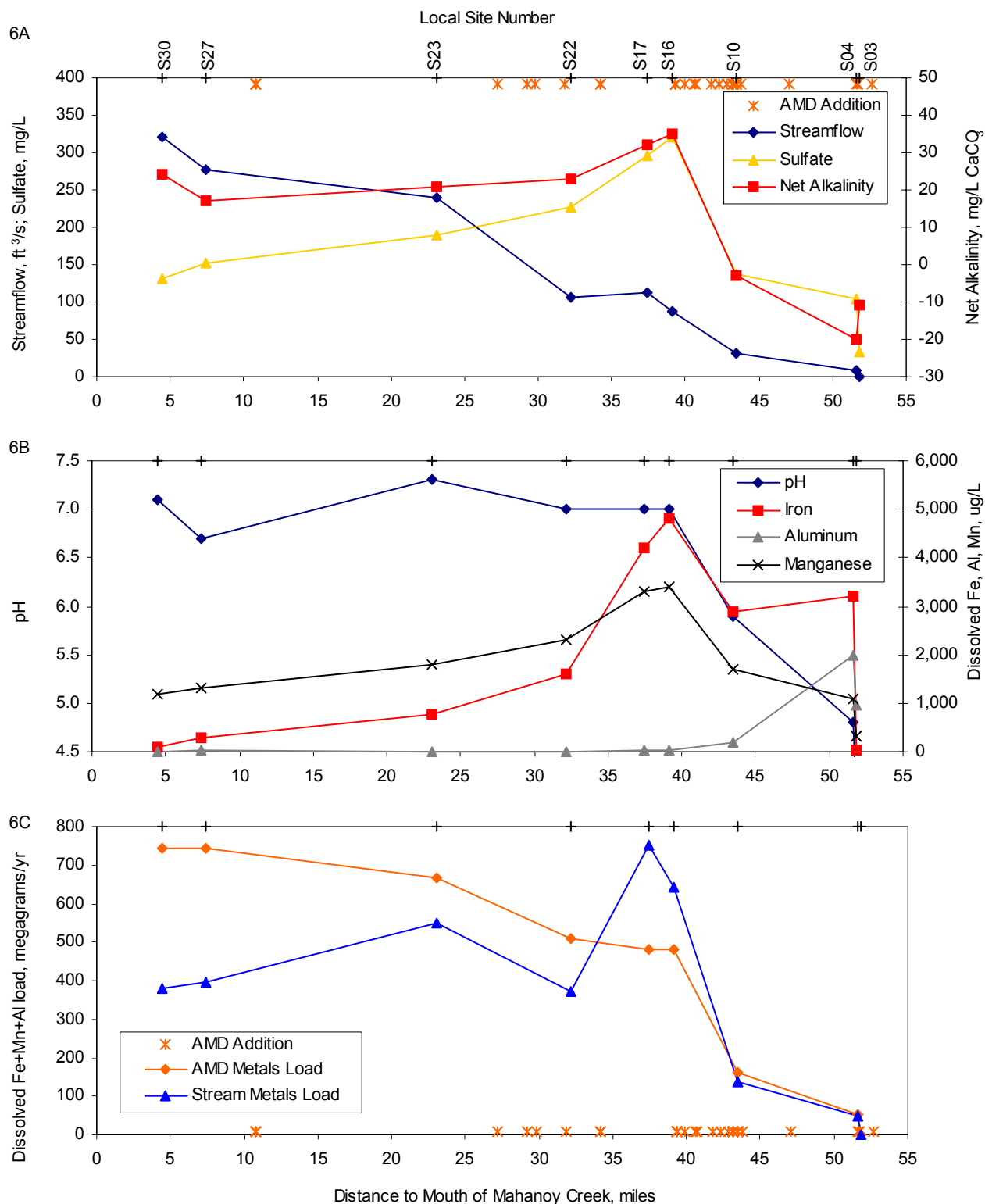


Figure 6. Streamflow and water quality at sites along the main stem of Mahanoy Creek, Pennsylvania, March 2001: (A) Streamflow and concentrations of sulfate and net alkalinity; (B) pH and concentrations of dissolved iron, aluminum, and manganese; and (C) loading of metals within stream and summed from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

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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 ft³/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 ft³/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 ft³/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 ft³/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 m²) immediately upstream of S14. In March and August 2001, flows at S14 were 12.1 and 5.5 ft³/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 ft³/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 ft³/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 equaled) 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 ft³/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 ft³/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 ft³/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 ft³/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 ft³/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 ft³/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 dis-

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charges 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 ft³/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 ft³/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 ft³/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 ft³/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 ft³/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 mi² (58.8 km²) 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 ft³/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 mi² (7.3 km²) 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 ft³/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 ft³/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 ft³/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 ft³/s (232,800 L/min) and during August 21-23, 2001, was 30 ft³/s (50,970 L/min), bracketing the long-term average streamflow of 85.1 ft³/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.

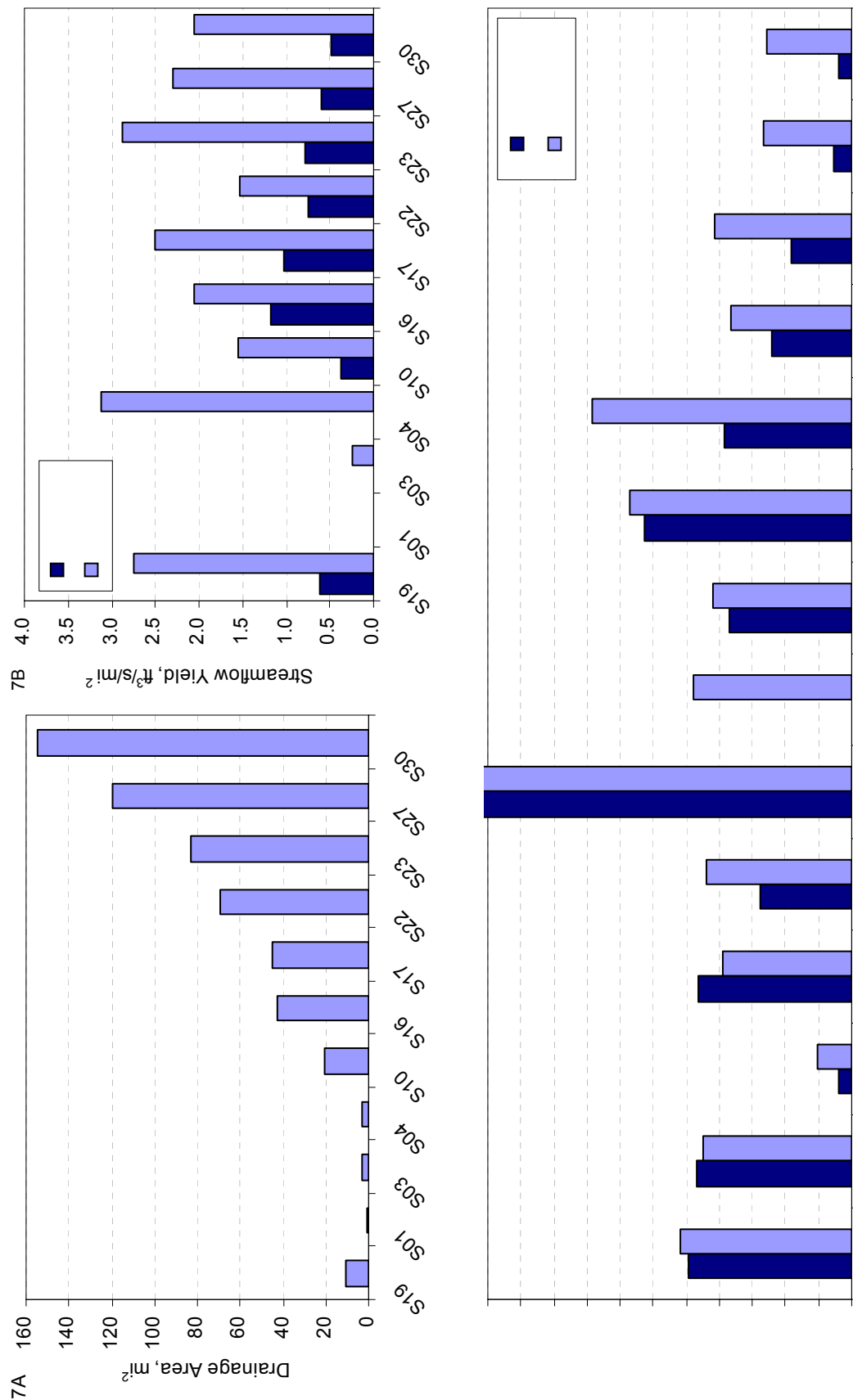


Figure 7. (A) Contributing area in square miles (mi²), (B) streamflow yields in cubic feet per second per square mile (ft³/s/mi²), and (C) dissolved metals loading percentage (%) of abandoned mine drainage (AMD) metals loading at streamwater monitoring sites in the Mahanoy Creek Basin, Pennsylvania, 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 mi² (58.8 km²) 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

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Table 4. Fish species identified and number of individuals counted during ecological survey of Mahanoy Creek and selected tributary streams, Pennsylvania, October 2001¹

Taxa		Mini- mum pH in Pa. ²	Pollu- tion toler- ance ³	Mahanoy Creek				Schwa- ben Cr nr Red Cross (S28)
ORDER Family <i>Genus species</i>	Common name			Girard- ville (S10)	Ashland (S16)	Gowen City (S23)	Kneass (S30)	
CYPRINIFORMES								
Cyprinidae								
<i>Campostoma anomalum</i>	Stoneroller	6.0	M	0	0	0	4	36
<i>Cyprinella analostana</i>	Spotfin shiner	6.4	M	0	0	1	9	308
<i>Exoglossum maxillingua</i>	Cutlips minnow	6.1	I	0	0	0	0	12
<i>Luxilus cornutus</i>	Common shiner	6.0	M	0	0	0	0	4
<i>Nocomis micropogon</i>	River chub	6.0	I	0	0	1	0	0
<i>Notropis hudsonius</i>	Spottail shiner	6.4	M	0	0	2	232	350
<i>Notropis procne</i>	Swallowtail shiner	6.5	I	0	0	0	4	0
<i>Notropis rubellus</i>	Rosyface shiner	6.0	I	0	0	0	8	21
<i>Pimephales notatus</i>	Bluntnose minnow	5.6	T	0	0	0	13	40
<i>Pimephales promelas</i>	Fathead minnow	6.5	T	1	0	0	0	0
<i>Rhinichthys atratulus</i>	Blacknose dace	5.6	T	0	0	39	0	1
<i>Rhinichthys cataractae</i>	Longnose dace	5.9	I	0	0	3	0	15
<i>Semotilus atromaculatus</i>	Creek chub	5.2	T	0	4	11	0	15
<i>Semolitus corporalis</i>	Fallfish	6.1	M	0	0	2	0	24
Catostomidae								
<i>Catostomus commersoni</i>	White sucker	4.6	T	19	18	51	40	23
<i>Hypentelium nigricans</i>	Northern hog sucker	6.0	I	0	0	0	0	1
SILURIFORMES								
Ictaluridae								
<i>Noturus insignis</i>	Margined madtom	5.9	M	0	0	0	0	5
SALMONIFORMES								
Salmonidae								
<i>Salmo trutta</i>	Brown trout	5.9	M	0	0	2	0	0
PERCIFORMES								
Centrarchidae								
<i>Ambloplites rupestris</i>	Rock bass	6.0	M	0	0	0	4	6
<i>Lepomis cyanellus</i>	Green sunfish	6.4	T	0	0	1	3	0
<i>Micropterus dolomieu</i>	Smallmouth bass	6.0	M	0	0	9	163	60
Percidae								
<i>Etheostoma olmstedi</i>	Tessellated darter	5.9	M	0	0	9	19	16
<i>Etheostoma blennioides</i>	Greenside darter	6.0	M	0	0	3	24	34
<i>Etheostoma zonale</i>	Banded darter	6.0	I	0	0	0	21	31
<i>Percina peltata</i>	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 unclaimed 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 ($\text{Fe}(\text{OH})_3$), schwertmannite ($\text{Fe}_8\text{O}_8(\text{OH})_6\text{SO}_4$), 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 ($\text{KAl}_3\text{Si}_3\text{O}_{10}(\text{OH})_2$), kaolinite ($\text{Al}_2\text{Si}_2\text{O}_5(\text{OH})_4$), and gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) were locally present at the AMD sites. Downstream from the AMD sources, streambed sediment samples were composed of quartz, muscovite, kaolinite, chlorite ($\text{Mg}_5\text{Al}_2\text{Si}_3\text{O}_{10}(\text{OH})_8$), 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.

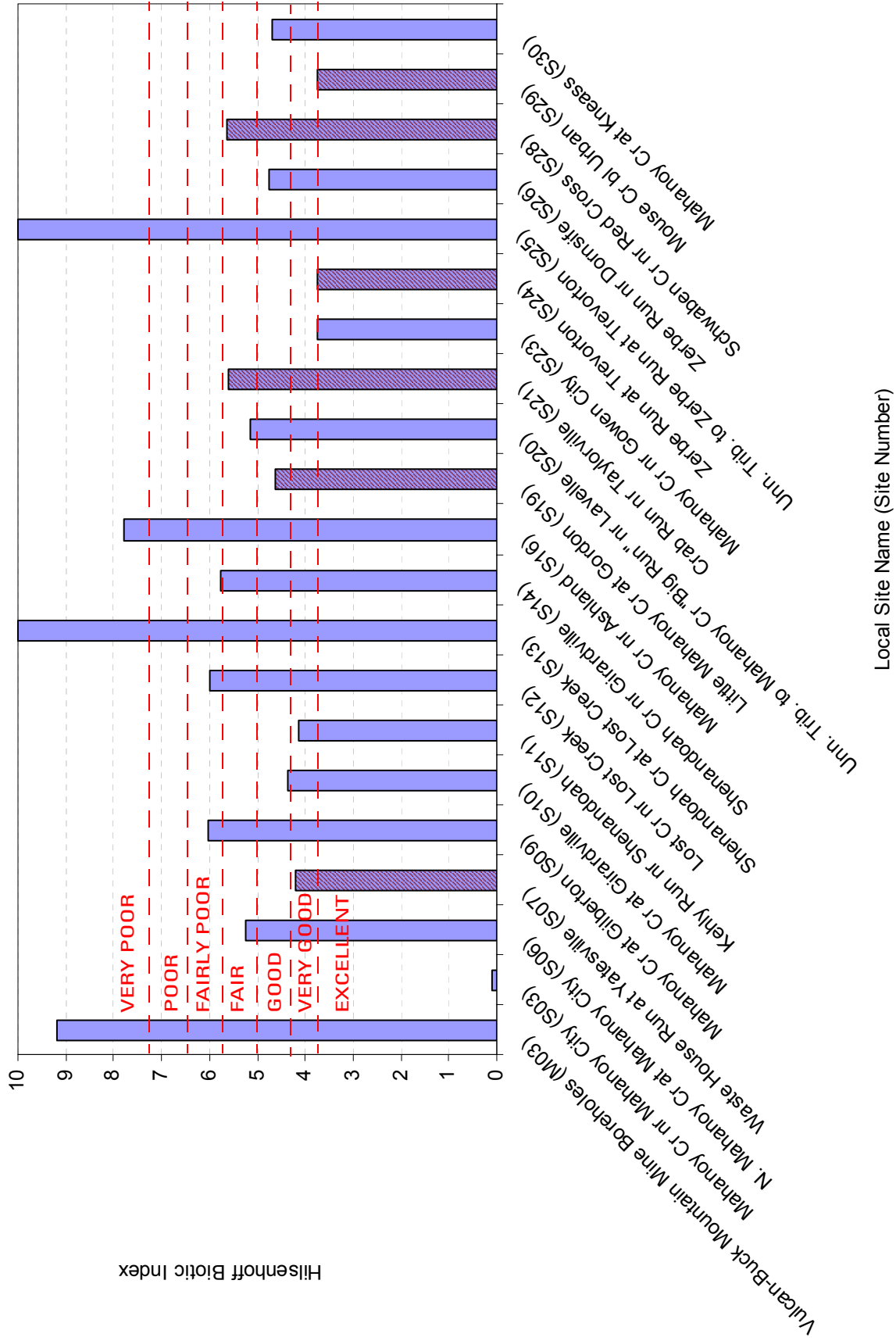


Figure 8. Hilsenhoff (1988) biotic index of aquatic macroinvertebrate diversity, 2000-2002, at sites sampled for water chemistry in the Mahanoy Creek Basin, Pennsylvania. Diagonal-patterned bars indicate unmined subbasins. Local site numbers shown in figure 1. Macroinvertebrates collected and identified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

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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 ($\mu\text{g/g}$) 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			1978-79	1980-81
		Dry	Wet	Dry	Wet	Dry	Wet			
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.

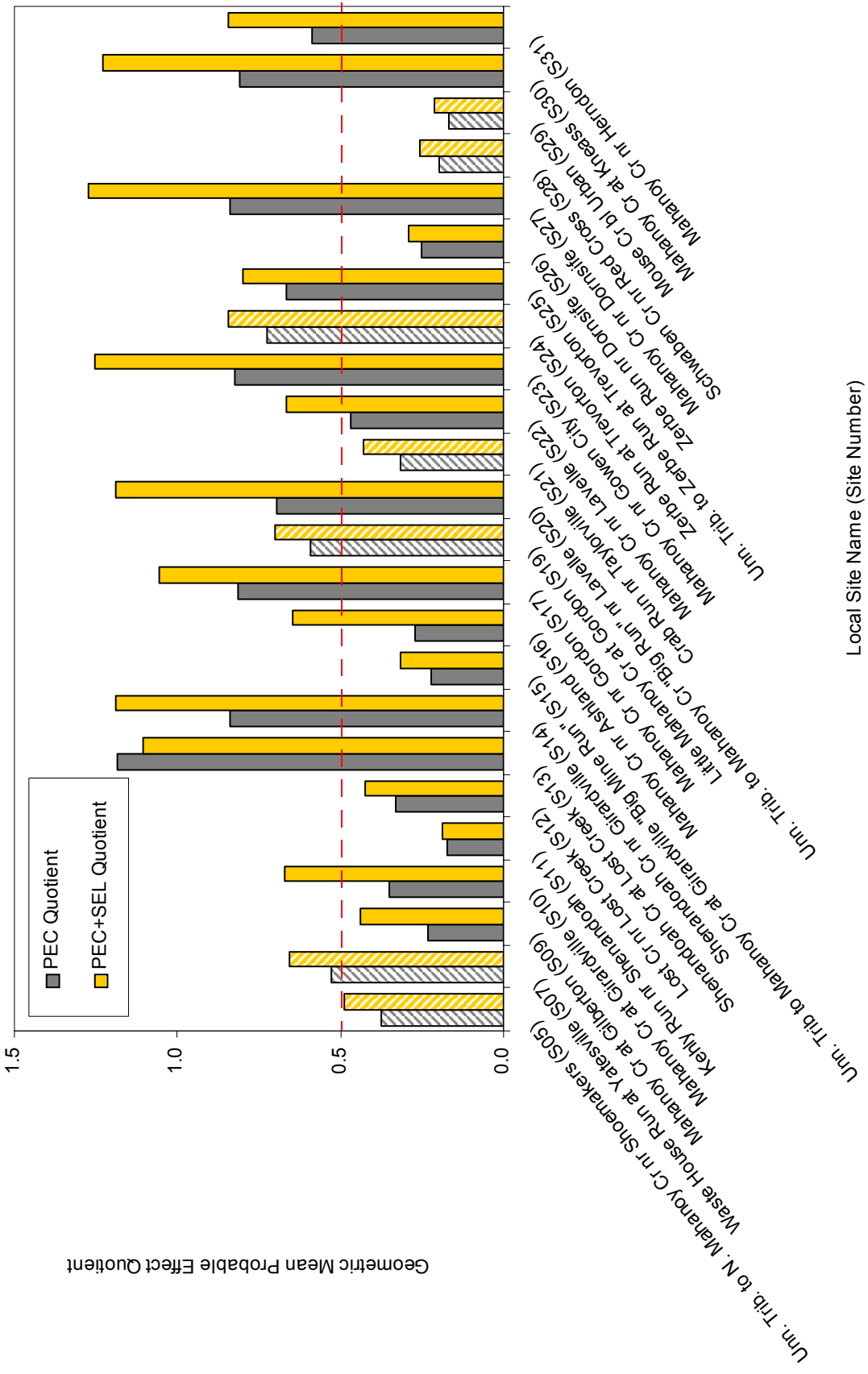


Figure 9. Geometric mean of the ratios of observed and probable effect concentrations (PECs) or PECs plus severe effect levels (SELs) for metals in streambed sediments in the Mahanoy Creek Basin, Pennsylvania, August 2001. Local site numbers shown in figure 1. Diagonal-patterned bars indicate unmined 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.

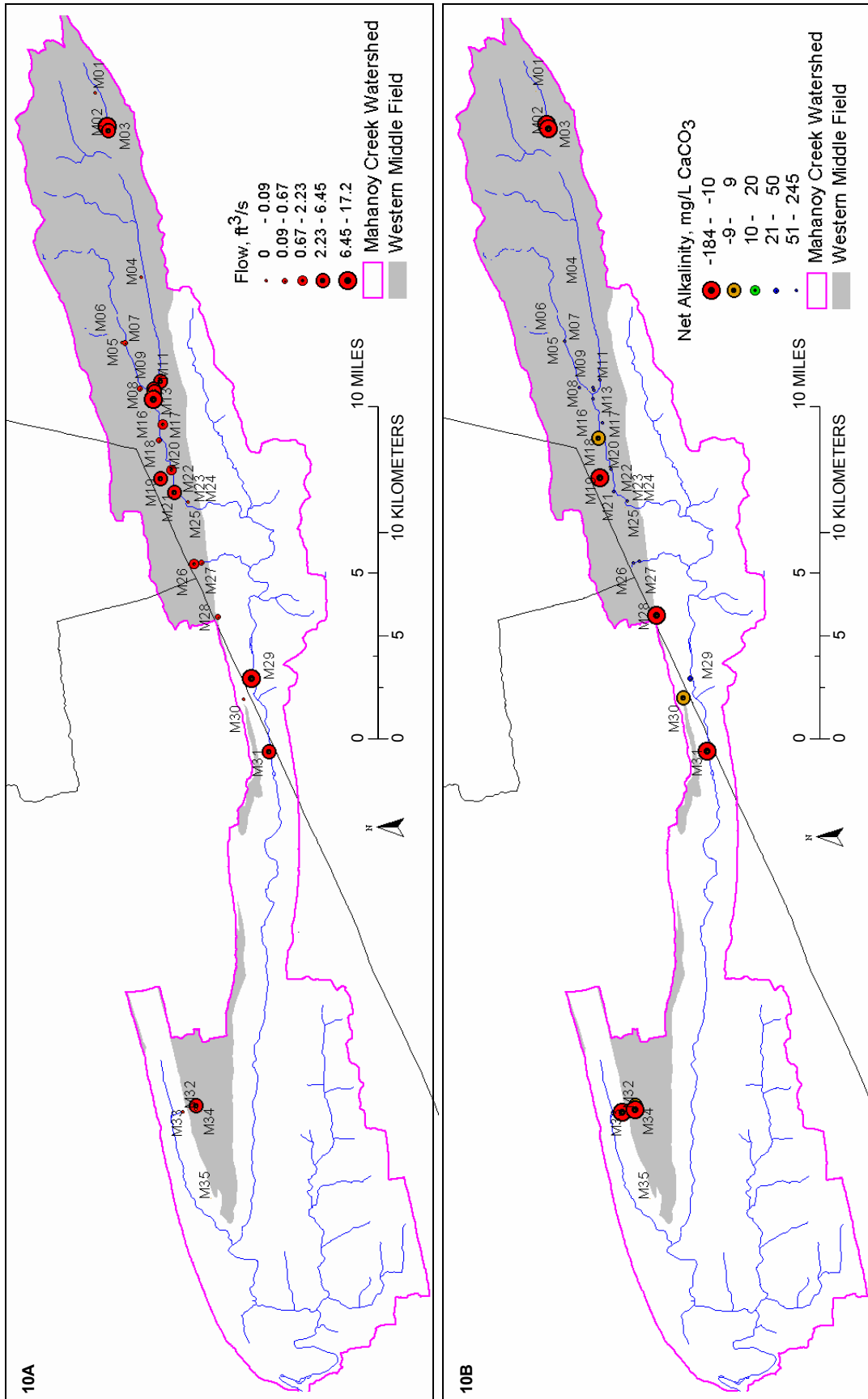


Figure 10. Mahanoy Creek Basin, Pennsylvania, showing abandoned mine drainage locations and corresponding (A) flow rate, (B) net alkalinity concentration, and loading of dissolved metals: (C) iron, (D) manganese, (E) aluminum, and (F) iron+manganese+aluminum, March 2001. Flow rate in cubic feet per second (ft³/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L CaCO₃); loading in megagrams per year (Mg/yr).

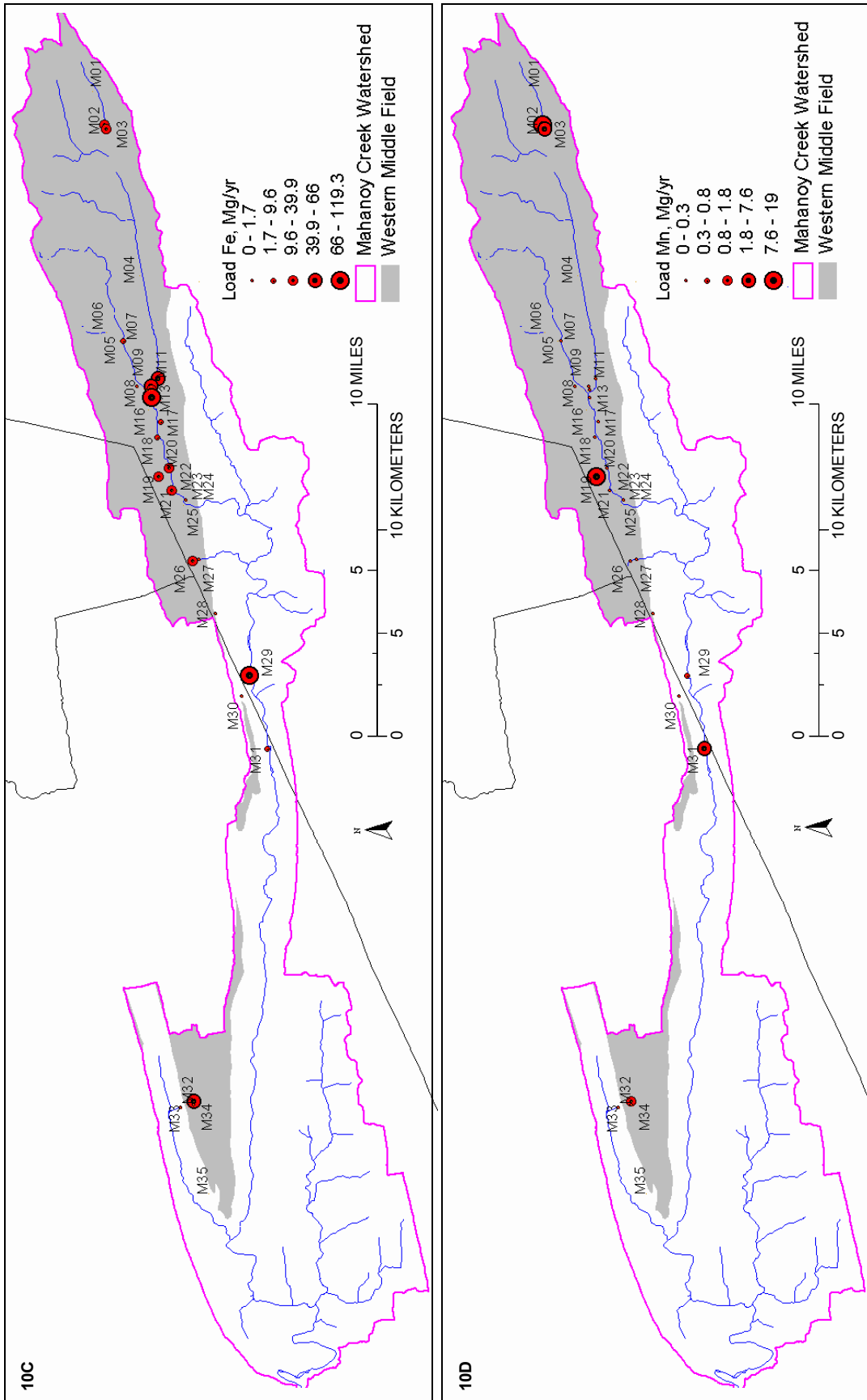


Figure 10. Mahanoy Creek Basin, Pennsylvania, showing abandoned mine drainage locations and corresponding (A) flow rate, (B) net alkalinity concentration, and loading of dissolved metals: (C) iron, (D) manganese, (E) aluminum, and (F) iron+manganese+aluminum, March 2001. Flow rate in cubic feet per second (ft³/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L CaCO₃); loading in megagrams per year (Mg/yr).—Continued

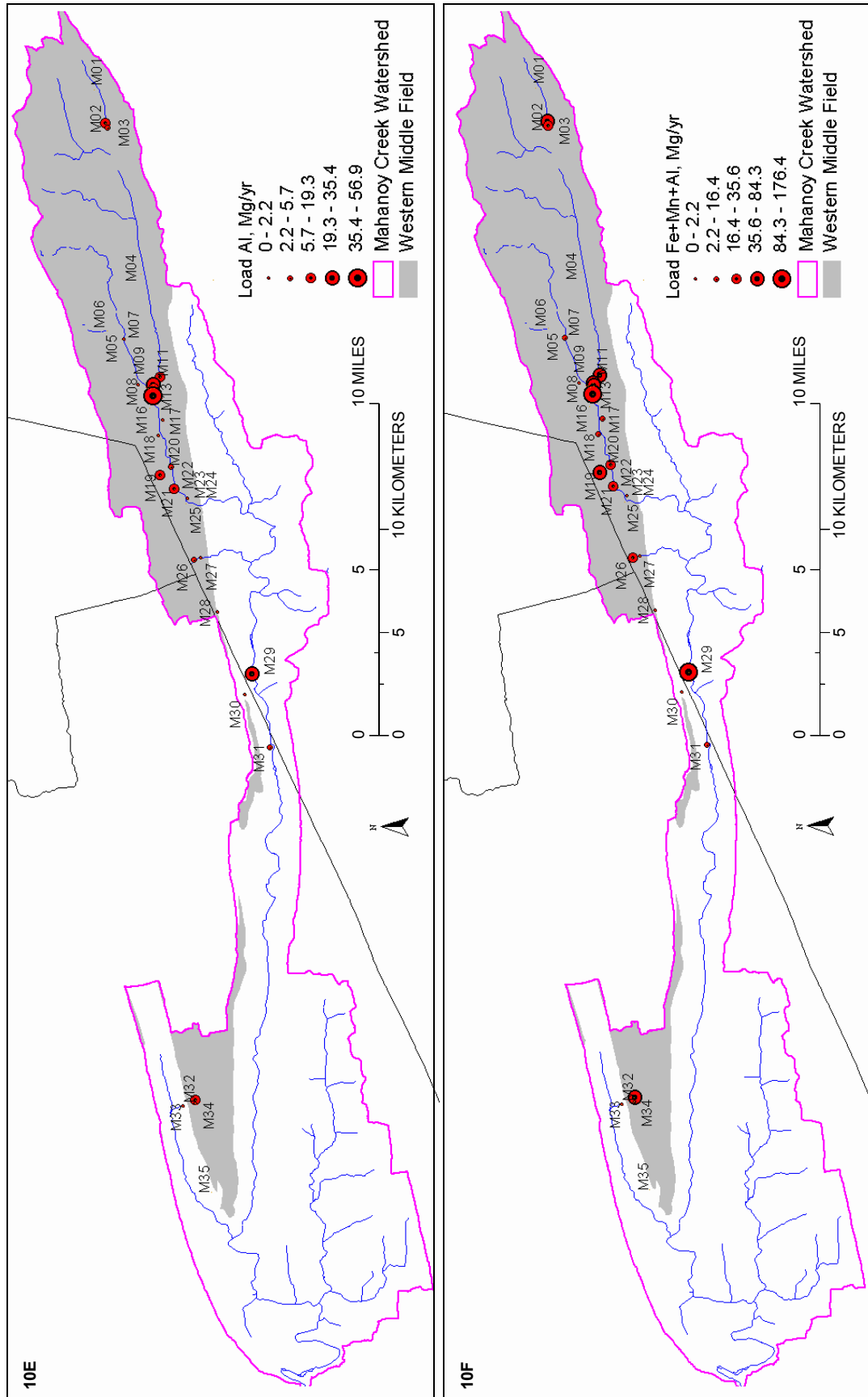


Figure 10. Mahanoy Creek Basin, Pennsylvania, showing abandoned mine drainage locations and corresponding (A) flow rate, (B) net alkalinity concentration, and loading of dissolved metals: (C) iron, (D) manganese, (E) aluminum, and (F) iron+manganese+aluminum, March 2001. Flow rate in cubic feet per second (ft³/s); net alkalinity concentration in milligrams per liter as calcium carbonate (mg/L CaCO₃); 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 ft³/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 ft³/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 ft³/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 ft³/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-mi² (29.3-km²) 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 ft³/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 ft³/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 ft³/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 ft³/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-mi² (29.3 km²) 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 ft³/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 ft³/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 ft³/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 mi² (10.4 km²) 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 ft³/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 ft³/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 ft³/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 ft³/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 ft³/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 ft³/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 ft³/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 ft³/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 (table 3). 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 ft³/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 ft³/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 dis-

charges 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 mi² (18.1 km²) 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 ft³/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 ft³/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 ft³/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 ft³/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 mi² (4.8 km²) 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 ft³/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 ft³/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 ft³/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, alumi-

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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)

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Appendix D.

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