

Effects of Historical Coal Mining and Drainage from Abandoned Mines on Streamflow and Water Quality in Newport and Nanticoke Creeks, Luzerne County, Pennsylvania, 1999–2000

In cooperation with the Earth Conservancy

Scientific Investigations Report 2007-5061

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Conversion Factors, Abbreviated Water-Quality Units, and Datum

Multiply	By	To obtain
Length		
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
Area		
acre	4,047	square meter (m ²)
acre	0.4047	hectare (ha)
square mile (mi ²)	2.590	square kilometer (km ²)
Pressure		
atmosphere, standard (atm)	101.3	kilopascal (kPa)
Flow rate		
cubic foot per second (ft ³ /s)	0.02832	cubic meter per second (m ³ /s)
cubic foot per second (ft ³ /s)	448.9	gallon per minute
cubic foot per second per square mile [(ft ³ /s)/mi ²]	0.01093	cubic meter per second per square kilometer [(m ³ /s)km ²]
Mass		
pound, avoirdupois (lb)	0.4536	kilogram (kg)
ton, short (2,000 lb)	0.9072	megagram (Mg)
Iron-removal rate		
pounds per acre per day [(lb/acre)/d]	0.1121	grams per square meter per day [(g/m ²)/d]

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:
 $^{\circ}\text{F} = (1.8 \times ^{\circ}\text{C}) + 32$

Vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29).

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

Altitude, as used in this report, refers to distance above the vertical datum.

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

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter ($\mu\text{g/L}$).

Concentrations of total coliform and *Escherichia coli* bacteria in water are reported in colonies per 100 milliliters (col/100 mL)

Other abbreviations:

mol/L moles per liter

$\text{L}^2/(\text{mol}^2 \times \text{atm} \times \text{s})$ square liters per square mole per atmosphere per second

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Abstract

This report characterizes the effects of historical mining and abandoned mine drainage (AMD) on streamflow and water quality and evaluates potential strategies for AMD abatement in the 14-square-mile Newport Creek Basin and 7.6-square-mile Nanticoke Creek Basin. Both basins are mostly within the Northern Anthracite Coal Field and drain to the Susquehanna River in central Luzerne County, Pa. The U.S. Geological Survey (USGS), in cooperation with the Earth Conservancy, conducted an assessment from April 1999 to September 2000 that included (1) continuous stage measurement at 7 sites; (2) synoptic water-quality and flow sampling at 21 sites on June 2-4, 1999, and at 24 sites on October 7-8, 1999; and (3) periodic measurement of flow and water quality at 26 additional sites not included in the synoptic sampling effort.

Stream water and surface runoff from the unmined uplands drain northward to the valley, where most of the water is intercepted and diverted into abandoned underground mines. Water that infiltrates into the mine workings becomes loaded with acidity, metals, and sulfate and later discharges as AMD at topographically low points along lower reaches of Newport Creek, Nanticoke Creek, and their tributaries. Differences among streamflows in unmined and mined areas of the watersheds indicated that (1) intermediate stream reaches within the mined area but upgradient of AMD sites generally were either dry or losing reaches, (2) ground water flowing to AMD sites could cross beneath surface-drainage divides, and (3) AMD discharging to the lower stream reaches restored volumes lost in the upstream reaches.

The synoptic data for June and October 1999, along with continuous stage data during the study period, indicated flows during synoptic surveys were comparable to average values. The headwaters upstream of the mined area generally were oxygenated (dissolved oxygen range was 4.7 to 11.0 mg/L [milligrams per liter]), near-neutral (pH range was 5.8 to 7.6), and net alkaline (net alkalinity range was 2.0 to 25.0 mg/L CaCO₃), with relatively low concentrations of sulfate (6.40 to

24.0 mg/L) and dissolved metals (less than 500 µg/L [micrograms per liter] of iron, manganese, and aluminum). In contrast, the AMD discharges and downstream waters were characterized by elevated concentrations of sulfate and dissolved metals that exceeded Federal and State regulatory limits.

The largest AMD sources were the Susquehanna Number 7 Mine discharge entering Newport Creek near its mouth (flow range was 4.7 to 19 ft³/s [cubic feet per second]), the Truesdale Mine Discharge (Dundee Outfall) entering Nanticoke Creek about 0.5 mile upstream of Loomis Park (flow range was 0.00 to 38 ft³/s), and a mine-pit overflow entering near the midpoint of Newport Creek (flow range was 4.0 to 6.9 ft³/s). The three large discharges were poorly oxygenated (dissolved oxygen concentration range was <0.05 to 6.4 mg/L) and had elevated concentrations of sulfate (range was 710 to 890 mg/L) and low concentrations of dissolved aluminum (less than 25 µg/L), but they had distinctive concentrations of net alkalinity and dissolved iron and manganese. Effluent from the Susquehanna Number 7 Mine was near-neutral (pH range was 5.9 to 6.6) and net alkaline (net alkalinity range was 12.0 to 42.0 mg/L CaCO₃) with elevated concentrations of sulfate (718 to 1,170 mg/L), dissolved iron (52,500 to 77,400 µg/L), and manganese (5,200 to 5,300 µg/L). Effluent from the Truesdale Mine also was near-neutral (pH range was 5.9 to 6.3) but had variable net alkalinity (-19.0 to 57.0 mg/L CaCO₃) with elevated concentrations of sulfate (571 to 740 mg/L), dissolved iron (30,500 to 43,000 µg/L), and manganese (3,600 to 5,200 µg/L). Effluent from the mine-pit overflow in Newport Creek Basin was acidic (pH range was 4.3 to 5.0; net alkalinity range was -42 to -38 mg/L CaCO₃) with elevated concentrations of sulfate (800 to 840 mg/L), iron (13,000 to 16,000 µg/L), and manganese (6,800 to 7,000 µg/L). Although the three large AMD sources did not contain detectable concentrations of dissolved aluminum, a small AMD source in the Nanticoke Creek Basin (flow less than 0.01 ft³/s to 0.06 ft³/s), along with other small AMD sources entering the South Branch of Newport Creek between Wanamie and Sheatown (flows less than 0.89 ft³/s), had elevated concentrations of dissolved aluminum (3,100 to 38,600 µg/L) that exceeded criteria for protection of aquatic organisms.

The chemistry of stream water after mixing with AMD inputs was variable, depending on the relative quantities of

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AMD and other water sources. For example, decreased flow rates and net alkalinities of AMD from the Truesdale Mine coupled with increased acid production from more extensive iron hydrolysis within Nanticoke Creek could explain the acidic quality about 0.5 mile downstream of the mine during drought (minimum pH was 3.2) compared to the near-neutral quality during normal flows (median pH was 6.8). Other co-occurring influences such as alkalinity from intermittent sewage inflows could explain the bimodal pH distribution near the mouth of South Branch; water had near-neutral pH (pH greater than 6.0) when sewage was abundant but acidic pH (less than 4.5) when it was not.

AMD in other mined areas with chemistry and flow similar to the discharges sampled for this study, has been treated with passive strategies that may include amendment of influent chemistry and almost always include retention in aerobic wetlands. For water similar in quality and quantity to the Truesdale Mine Discharge, with iron loading rates approaching 326 kilograms per day, aerobic wetlands of 4 to 16 acres combined with an alkalinity source have been used for passive treatment. For large, consistently net alkaline flows, such as the AMD discharge from the Susquehanna Number 7 Mine, wetlands of 12 to 49 acres have been used to remove dissolved iron, without supplemental alkalinity, provided that pH is maintained near neutrality. AMD sources with large flow rates, low pH, and elevated concentrations of dissolved metals, such as the mine-pit overflow, commonly warrant active treatment. For example, efficient alkalinity-producing systems, such as lime dosing, followed by ponds or wetlands of approximately 3 to 13 acres have been used to neutralize AMD and remove dissolved iron for similar situations.

Introduction

Losses of stream water to and contaminated drainage from abandoned anthracite mines impair the uses of Newport Creek and Nanticoke Creek, which drain adjacent 14-mi² and 7.6-mi² basins to the Susquehanna River in central Luzerne County, Pa. (fig. 1). The Newport and Nanticoke Creek Basins are underlain by the Northern Anthracite Coal Field, also known as the Wyoming Basin (Ladwig and others, 1984; Hollowell, 1971). More than 100 years of underground and surface mining for coal have contaminated or disrupted local water supplies (Skelly & Loy, Inc., 1974; GEO-Technical Services, Inc., 1975; Growitz and others, 1985; Wood, 1996). Extensive areas of spoil and mine refuse have not been reclaimed, and the abandoned underground workings are interconnected and substantially flooded, creating an underground mine pool. Collapse of underground workings has caused subsidence on the surface, where surface flow can be intercepted and diverted into the underground mines. Water that encounters pyrite and its oxidation products in the underground mines and associated spoil can become loaded with acidity, iron, and sulfate. In downstream reaches, contaminated ground water resurfaces as abandoned mine

drainage (AMD) from fractures or mine openings at topographically low points. The AMD contaminates stream water but also contributes substantially to base flow.

Studies in the mid-1970s considered prevention of surface-water leakage as the primary strategy for rehabilitation of Newport Creek Basin and conventional (active) treatment of AMD as the best approach to address water-quality degradation in Nanticoke Creek Basin (Skelly & Loy, Inc., 1974; GEO-Technical Services, Inc., 1975). Stream-channel lining and relocation, regrading and revegetating strip mines and refuse piles, and channel excavation were among the measures recommended to reduce AMD volume in Newport Creek Basin by preventing stream-water leakage into the mines (Skelly & Loy, Inc., 1974). However, remaining AMD would have had to be treated because the 1974 abatement plan for Newport Creek Basin, if implemented, would have removed only 10 percent of the loads of iron and acidity in the basin. In Nanticoke Creek Basin, treatment of AMD with hydrated lime was proposed (GEO-Technical Services, Inc., 1975) but was not economically feasible at the time because of the infrastructure, chemical amendments, and operational costs associated with this active-treatment approach.

In the mid-1990s, passive treatments such as limestone drains and aerobic wetlands were developed as low-cost alternatives to conventional treatment to remediate AMD (Hedin and others, 1994). Small-scale passive treatment has been attempted at two AMD discharges in the Nanticoke Creek Basin (Klemow, 2002b, 2003) and is now (2007) viewed as a viable option to ameliorate AMD contamination on a larger scale. Passive-treatment alternatives are selected on the basis of hydrologic and chemical properties of the contaminated influent. Because the range and variability of these properties influence selection of an appropriate treatment strategy, monitoring water quality over a range of flows generally is necessary to identify potential treatment or other pollution-abatement strategies.

From 1999 to 2000, the U.S. Geological Survey (USGS), in cooperation with Earth Conservancy (EC) and in collaboration with Wilkes University (WU), characterized the chemical and hydrologic properties of stream water and AMD discharges in Newport and Nanticoke Creek Basins. Measurements of flow and quality of stream water and AMD were completed throughout the study area over a range of hydrologic conditions. These data were then used to characterize the effects of historical mining and AMD and to identify potential strategies for remediation of AMD. The data collected for this study represent the type of baseline information needed for documentation of water-quality changes following passive treatment of mine drainage in Pennsylvania and other similar hydrogeologic settings. Because many AMD discharges throughout the Anthracite Coal Region (fig. 1) are similar in quality and quantity to those sampled for this investigation (Growitz and others, 1985; Wood, 1996), potential remedial strategies presented here may be transferable with little adaptation.

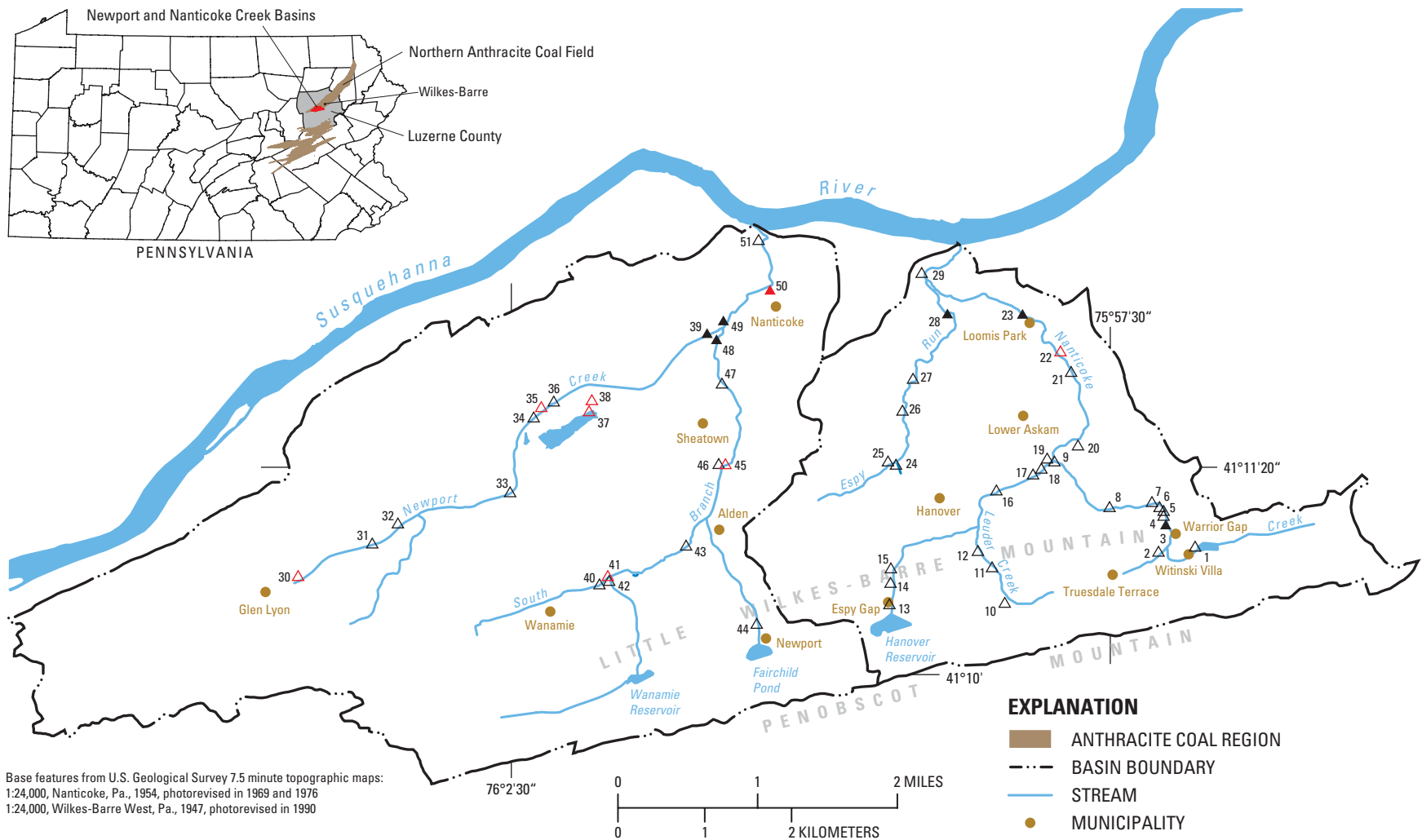


Figure 1. Location of Newport and Nanticoke Creek Basins and the associated monitoring network.

Purpose and Scope

This report outlines a process to assess remediation strategies for AMD in Newport and Nanticoke Creek Basins and other mined areas. The effects of historical mining and drainage from abandoned coal mines on the streamflow and water quality are presented along with potential remediation strategies for AMD sources to Newport Creek, Nanticoke Creek, and their tributaries. Geochemical modeling was used to evaluate the effect of iron oxidation and hydrolysis on the quality of water from the Truesdale Mine Discharge (locally referred to as the Dundee Outfall) in Nanticoke Creek Basin and to simulate changes in pH and concentration of dissolved iron that could result with treatment of the AMD effluent.

Hydrologic and chemical data were collected throughout Newport and Nanticoke Creek Basins from April 1999 to September 2000. Field data for instantaneous flow rate, temperature, specific conductance (SC), dissolved oxygen (DO), pH, oxidation-reduction potential (Eh), alkalinity, and acidity were collected simultaneously with water samples at a total of 51 AMD and stream sites. Streamflows at seven sites were gaged (sites 3, 23, and 28 in Nanticoke Creek Basin and 39, 48, 49, and 50 in Newport Creek Basin; fig. 1) from April 1999 through March 2000, allowing assessment of the spatial variability of flow and correlation with water-quality data collected by USGS and WU. Water-quality samples were analyzed for major cations and anions and, at some sites, nutrients and bacteria concentrations.

Description of the Study Area

Newport and Nanticoke Creeks drain a combined area of 21.6 mi² that is mostly within the Northern Anthracite Coal Field in Luzerne County, Pa. The headwaters of each stream flow northward from Penobscot and Little Wilkes-Barre Mountains through areas intensely mined by surface and underground methods before joining the Susquehanna River below Wilkes-Barre, Pa. (fig. 1). The stream locations on figure 1 are shown as they appear on available topographic quadrangles (U.S. Geological Survey, 1976, 1990). However, substantial lengths of the streams lose water and can be dry during much of the year, and some stream segments have been rerouted. Mean annual precipitation in the study area is 36.9 in. based on 30 years of record at Wilkes-Barre/Scranton International Airport, which is located approximately 20 mi northeast of the study area (Northeast Regional Climate Center, 2006). Monthly mean temperature ranges from -3.2 °C in January to 22.3 °C in July (Northeast Regional Climate Center, 2006).

Topography in the study area has been altered by past mining activity, leaving refuse piles, crop falls, and strip pits scattered throughout each basin. Open strip mines now cover only about 7 percent of the study area, but many former strip mines are revegetated with second growth forest. These areas, along with forests growing where the land surface was not mined, cover 65 percent of the study area. Residential and commercial/

industrial development account for 22 percent of land use. The balance of land area is used for agricultural purposes (4 percent) or is undeveloped wetlands and open water (2 percent) (U.S. Geological Survey, 1996).

Geology and Mining History

Newport and Nanticoke Creek Basins are in the southern tip of the Anthracite Valley Section of the Ridge and Valley Physiographic Province (Sevon, 2000). The Anthracite Valley Section is a crescent-shaped synclinal basin that encompasses the tightly folded and faulted strata of the Northern Anthracite Coal Field. The coal field extends for approximately 62 mi in a southwest to northeast orientation (fig. 1). The maximum width is about 5 mi but narrows to roughly 3.5 mi in the study area (see Bergin, 1976, in appendix 1). Structurally, the valley pitches eastward from the western end and southwestward from the northeast end, with the axis of the syncline near the center (Itter, 1959). In the study area, the Truesdale Mine Discharge (site 22) is near the axis of the syncline (fig. 1).

The principal rock units underling Newport and Nanticoke Creek Basins and the adjoining areas of the coal field are the Mauch Chunk, Pottsville, and Llewellyn Formations (appendix 1). The Mauch Chunk Formation is of Late Mississippian and Early Pennsylvanian age and is successively overlain by the Pottsville and Llewellyn Formations of Pennsylvanian age. The Pottsville and Llewellyn Formations are characterized by tightly folded sedimentary rock units that commonly are interrupted by faults. Faults are most numerous in the Llewellyn Formation, but many extend into the underlying Pottsville Formation (Bergin, 1976) and may facilitate mixing of ground water. The Pottsville and Mauch Chunk Formations form the high ridges of Little Wilkes-Barre and Penobscot Mountains (fig. 1), and the overlying, less resistant, Llewellyn Formation forms the valley floor.

The Mauch Chunk Formation ranges from 200 to 500 ft thick in the Anthracite Coal Region (Wood and others, 1986) and is divided into three informal units (the upper, middle, and lower members) that are difficult to differentiate because of the lack of stratigraphic markers coupled with extensive faulting and folding. Lithologies are characterized by red and gray conglomerate, sandstone, siltstone, and shale of varying thickness. Cementing media include silica, limonite, hematite, unidentified clay minerals, sericite, and calcite (Wood and others, 1969). The Pottsville Formation is relatively thin in the study area (less than 100 ft) compared to the other areas of the Anthracite Coal Region where the thickness is nearly 1,600 ft (Brady and others, 1998). Although this formation contains several mineable coal seams elsewhere in the Anthracite Coal Region (Wood and Trexler, 1968), none are present in the study area (Bergin, 1976). The dominant lithology is sandstone and pebble conglomerates. Pebbles in conglomerate and conglomeratic rocks are made up largely of vein quartz and quartzite. Cementing media include silica, sericite, unidentified clay minerals, and calcite (Wood and others, 1969).

The Llewellyn Formation overlies the Pottsville Formation and contains more than 20 economically important coal seams in the study area. On the basis of stratigraphic sections near the Truesdale Mine Discharge, the combined thickness of the coal seams is 90 to 100 ft (Bergin, 1976). The base of the Llewellyn Formation is the Lower Red Ash coal seam, which is over 10 ft thick and is as deep as 1,500 ft below NGVD 29 at the axis of the syncline (Bergin, 1976). The Lower Red Ash coal seam follows the general slope of the syncline and crops out along Little Wilkes-Barre Mountain near the headwaters of Newport and Nanticoke Creeks. Geologic cross sections showing the extent of underground mining in the study area and adjoining basins suggest that roughly 50 to 70 percent of coal in mineable seams was extracted (Bergin, 1976) prior to cessation of underground mine activities.

Unconsolidated glacial sediments consisting of gravel, sand, and clay, and mixtures of these materials overlie the Llewellyn Formation (Itter, 1959; Ash, 1950). These sediments are in and adjacent to the flood plain of the Susquehanna River, extending to a depth of up to 300 ft beneath the land surface and for a width of approximately 1 mi (Bergin, 1976). The average depth near Nanticoke, Pa., is 180 ft (Itter, 1959). These sediments were deposited during the Pleistocene Epoch, a period of alternating glacial advances and retreats that scoured the Wyoming Valley where the ice sheets moved parallel to the valley axis. Much of this over-deepened valley has since been filled with Pleistocene sediments that were carried by ice and fluvial forces. The valley commonly is referred to as the Buried Valley of the Susquehanna River (Itter, 1959).

The anthracite mining industry became economically important between 1825 and 1835 (Wood and others, 1969), owing to demand from a growing steel industry supplied by a new railroad network throughout eastern Pennsylvania, Delaware, New Jersey, and Maryland. These developments provided the market and infrastructure for large-scale coal-mining operations. In the study area, underground mining of anthracite coal began in the early 1800s and continued into the 1970s. Coal production from underground and surface-mining methods peaked at 35 million tons in 1924 but decreased shortly after and remained relatively constant until 1950, shortly after the end of World War II. With demand waning after the war, production began a steady decline after 1950 (Skelly and Loy, Inc., 1974).

The method of coal extraction in the study area was dictated by the proximity of a coal seam to the surface, the inclination or dip of the coal seam, and the available equipment for mining. Most coal underlying the study area was extracted from extensive underground mines that were developed by excavating “rooms” of coal and leaving “pillars” in place to support the overlying rock (Wood and others, 1969). This “room and pillar” approach relied largely on manual labor with picks, shovels, and mule-drawn carts during the 1800s and evolved to include diesel and electric-powered equipment in the 1900s. A drift-mining technique generally was used where multiple entries in close proximity could be excavated into an outcropping coal bed that dipped less than 20° from horizontal. Where the dip of

the coal bed was greater than 20°, shafts were excavated following the dip of the coal bed or vertically for access and extraction. Coal mined close to the surface sometimes resulted in the collapse of overlying strata, producing subsidence at the surface. Crop falls and other subsidence features presently indicate the locations of the Lower Red Ash (the lowest workable coal seam) and other coalbeds that crop out along the flanks of the Little Wilkes-Barre Mountain near the southern limit of the basins.

Once power shovels were developed, strip mining was utilized wherever coal was accessible at or near the land surface, mainly where coal beds crop out along the south-facing slope of Little Wilkes-Barre Mountain, near the Askam Syncline (Bergin, 1976; appendix 1), and various other places where coal seams are at or near the surface. Overburden (overlying non-marketable strata) was removed with power shovels and bulldozers and typically discarded on the downslope side. Coal extraction generally progressed until the mine depth restricted the use of machinery, rendering the method unprofitable. Most surface mines in the area were left open after mining ceased; some filled with water.

Hydrologic Setting

Underground mining of numerous coalbeds has resulted in a vast series of voids that are interconnected in many places but may be isolated in others. These voids and associated tunnels, drifts, and shafts are collectively referred to as mine workings. During active mining, water had to be kept out of the workings to allow access to the coal. This generally was accomplished by construction of flumes to minimize stream-water infiltration and pumping to expel water from the underground workings. For example, over 3,000 ft of Leuder Creek, a tributary of Nanticoke Creek (fig. 1), was flumed with concrete in 1958 to keep stream water from flowing into the Truesdale Mine workings (Dierks and others, 1962; fig. 2).

Presently (2007), water is no longer pumped from the abandoned underground mines and substantial volumes of the interconnected workings have flooded, creating an underground mine pool. Flumes are in disrepair (fig. 3), and subsidence of the ground surface is common leaving open pits with some having connections to the mine pool. Interception and diversion of surface flow into the underground mines was estimated in the mid-1970s to be 40 percent of the total annual precipitation (Skelly & Loy, Inc., 1974) but may now be greater because of further deterioration of concrete flumes (fig. 3). Extensive areas of spoil and refuse left from historical mining practices (fig. 4) have altered or blocked surface drainage and now facilitate infiltration of runoff and contamination.

The routes and rates of water flowing through the mine workings are complex and in many areas unknown (Hollowell, 1974). On a regional scale, ground-water recharge occurs at the valley sides and discharge points are near the valley floor. The difference in elevation head between the recharge areas and the discharge points promotes upward flow through hydraulically



Figure 2. View of a concrete flume built in 1958 to minimize infiltration of Leuder Creek into the underground mine workings, Luzerne County, Pennsylvania. Photograph from Dierks and others (1962).



Figure 4. Remnants of a flume built in 1958 to minimize infiltration of Leuder Creek into the underground mine workings, Luzerne County, Pennsylvania.



Figure 3. View of mine refuse blocking Nanticoke Creek upstream of Leuder Creek, Luzerne County, Pennsylvania.

connected strata in the mined area. The water eventually surfaces as AMD from topographically low shafts, fractures, boreholes, and seepage areas. A simplified schematic of conceptual flow routes based on known mine connections in 1974 is shown in figure 5 (Hollowell, 1974; GEO-Technical Services, Inc., 1975). Vertical and horizontal flow paths also are locally affected by barrier pillars forming the boundaries between the underground mines (fig. 5). As coal reserves were depleted, barrier pillars frequently were breached by mining (robbing) or subsidence (Ash, 1954), allowing water to seep from higher pools to lower ones (Hollowell, 1974). The elevation and extent of connections between adjacent mines and the location of downgradient AMD discharges control localized movement of

ground water through the mine workings (fig. 5). Topographic divides described by the surface-basin boundaries (fig. 1) serve to direct the flow of surface runoff but do not accurately depict ground-water divides because the mine workings extend beneath surface-basin boundaries. Because the mine workings underlay adjacent basins (fig. 5), interbasin transfer of surface and underground water is facilitated where stream water and surface runoff infiltrate into the workings.

The majority of water flowing downgradient through the mine workings discharges at the Truesdale Mine Discharge (site 22; fig. 6) in Nanticoke Creek Basin, at the Susquehanna Number 7 Mine discharge (site 50) near the mouth of Newport Creek, and at the South Wilkes-Barre Boreholes east of Warrior Run Basin (not sampled for this study). Underground flow paths are complicated by isolated pools above and away from the main pool, unusual recharge conditions, and variation in the location and extent of connections between the mines. As a result, the flow paths shown in figure 5 may not accurately represent the present flow paths within the study area and are used here only to provide a conceptual framework for interpretation of hydrologic and water-quality data collected during this study.

Water-Quality Degradation, Protection Standards, and Remediation Practices

Iron, aluminum, and to a lesser extent manganese are the primary metals that degrade water quality and impair aquatic uses in the study area. Iron is the principal contaminant in AMD and is derived mainly from oxidation of pyrite (FeS_2), a common mineral in coal-bearing strata (Stumm and Morgan, 1996). When the overall reaction for pyrite oxidation (eq. 1) proceeds to completion, pyrite, oxygen, and water react to form ferric hydroxide ($\text{Fe}(\text{OH})_3$ (s)), sulfate (SO_4^{2-}), hydrogen ions (H^+), and heat.

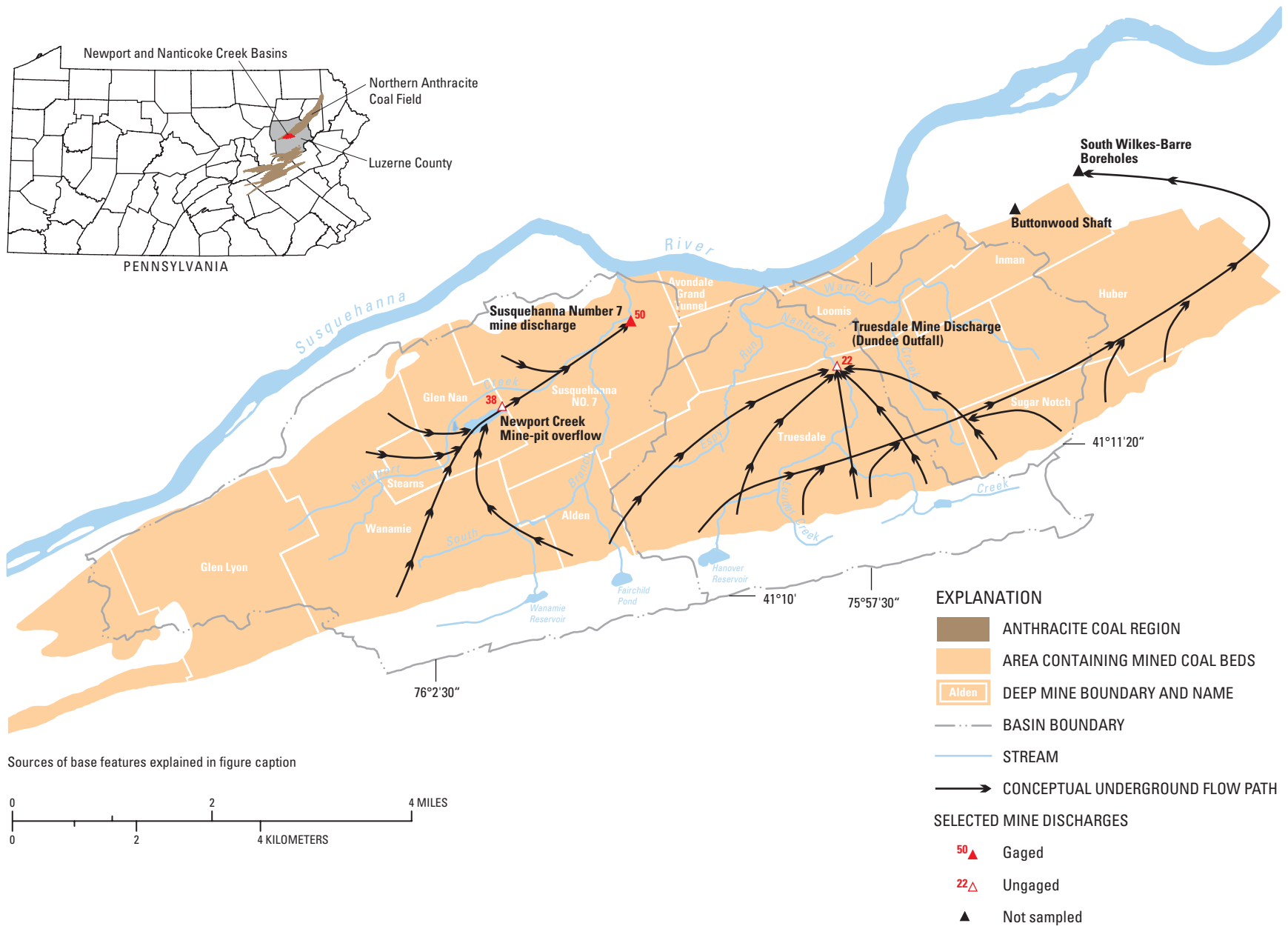


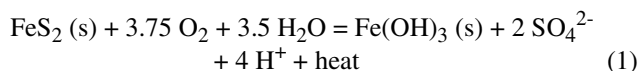
Figure 5. Deep-mine boundaries and conceptual underground flow paths toward mine discharges in Newport, Nanticoke, and Warrior Creek Basins, Luzerne County, Pennsylvania.

[Basin boundaries, streams, and reservoirs are from U.S. Geological Survey 7.5 minute, 1:24000 topographic maps: Nanticoke, Pa., 1954, photo revised in 1969 and 1976, and Wilkes-Barre West, Pa., 1947, photo revised in 1990. The area containing mined coalbeds and the deep mine names and boundaries are adapted from Ash (1954) and Hollowell (1974). Conceptual underground flow paths are adapted from hydrologic maps in Hollowell (1974) and GEO-Technical Services, Inc. (1975).

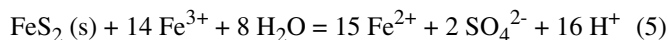
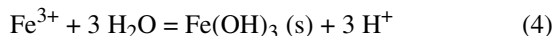
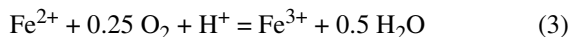
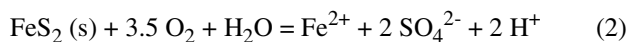
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Figure 6. Abandoned mine drainage discharging from the Truesdale Mine (Dundee Outfall) at site 22, Luzerne County, Pennsylvania.



The following reactions characterize various stages in the overall reaction (Stumm and Morgan, 1996):

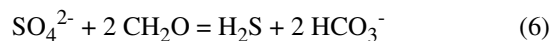


Oxidation of FeS_2 consumes dissolved oxygen and releases SO_4^{2-} , ferrous iron (Fe^{2+}), and H^+ into the water (eq. 2). As explained in more detail later in this report, dissolved Fe^{2+} can spontaneously oxidize to ferric iron (Fe^{3+}) (eq. 3), which is relatively insoluble. Most Fe^{3+} then hydrolyzes to form various amorphous oxide, hydroxide, or oxyhydroxysulfate precipitates, nominally designated here as $\text{Fe}(\text{OH})_3 (\text{s})$ (eq. 4).

$\text{Fe}(\text{OH})_3 (\text{s})$ can then settle from the water column and smother channel substrate, rendering stream reaches downstream of AMD discharges uninhabitable or inadequate for reproduction by many aquatic organisms. Dissolved Fe^{3+} can also react with $\text{FeS}_2 (\text{s})$, where SO_4^{2-} and Fe^{2+} are produced and Fe^{2+} may re-enter the reaction cycle via equation 3 (Stumm and Morgan, 1996).

Many large AMD sources in the Anthracite Region are suboxic to anoxic and contain alkalinity, SO_4^{2-} , and Fe^{2+} (Growitz and others, 1985; Wood, 1996). Generally, SO_4^{2-} is persistent in the aquatic environment, with little tendency for reaction. However, in the presence of organic carbon and in the absence of oxygen, SO_4^{2-} may be reduced by a bacterially

mediated reaction that produces hydrogen sulfide (H_2S) and bicarbonate (HCO_3^-) (eq. 6).



Sulfate reduction occurs when the water has been in contact with organic carbon long enough or reactive enough to deplete nearly all dissolved oxygen.

Acid released by the oxidation of FeS_2 (eqs. 2 and 5) and hydrolysis of Fe^{3+} (eq. 4) can dissolve various carbonate, oxide, and aluminosilicate minerals, releasing constituents such as manganese and aluminum. The mineral siderite (FeCO_3) may contribute iron but is notable as a source of manganese (Morrison and others, 1990; Cravotta and others, 1994). Manganese oxides also may be sources of manganese (Hem, 1985, p. 88-89). Other metals, particularly aluminum, can originate from the dissolution of aluminosilicate minerals (Rose and Cravotta, 1998).

Because of their capacity to consume alkalinity when precipitated as hydroxide or oxides, dissolved manganese and aluminum are recognized as potential sources of acidity in AMD (Kirby and Cravotta, 2004). Nevertheless, manganese does not readily oxidize at pH less than 8.0, and the toxicity of aluminum is a greater concern than its potential for acidity release. Aluminum is least soluble (and least toxic) between pH 5.7 to 6.2 where aluminum hydroxide precipitation limits concentrations to approximately 200 $\mu\text{g/L}$. Even at concentrations this low, reduction in survival and growth of Eastern Brook Trout and other fish has been documented (Baker and Schofield, 1982).

Various water- and sediment-quality guidelines are relevant for evaluating the quality of the AMD in mine discharges and downstream reaches on Newport and Nanticoke Creeks. Effluent from coal mines permitted after passage of Public Law 95-87, the Surface Mining Control and Reclamation Act of 1977 (SMCRA), must have alkalinity that exceeds its acidity concentration and must not have an instantaneous maximum concentration of iron, manganese, or aluminum that exceeds 7.0 mg/L, 5.0 mg/L, or 0.75 mg/L, respectively (Commonwealth of Pennsylvania, 1998a, 1998b). Additionally, the average daily concentration of sulfate must not exceed 250 mg/L for discharges that could affect public water supplies (Commonwealth of Pennsylvania, 1998a, 1998b). Although these effluent criteria generally are not enforced for discharges from coal mines in Pennsylvania that were abandoned before 1977, in-stream criteria for chemical constituents have been incorporated in recent “total maximum daily loads” (TMDLs) for mining-affected basins (Commonwealth of Pennsylvania, 2001a; Pennsylvania Department of Environmental Protection, 2002, 2004a). TMDLs can be considered to be a basin budget for pollutants, representing the total amount of pollutants that can be assimilated by a stream without causing water-quality standards to be exceeded. The criteria for TMDLs generally are consistent with those established to meet the warm-water fishery (WWF) or cold-water fishery (CWF) designation of a stream or other freshwater body (Commonwealth of Pennsylvania, 2002):

- temperature during July and August not to exceed 66°F (18.9°C) for CWF or 87°F (30.6°C) for WWF;
- DO concentration greater than 5.0 mg/L for CWF and 4.0 mg/L for WWF;
- alkalinity not less than 20 mg/L CaCO₃, except where natural conditions are less;
- pH not less than 6.0 or greater than 9.0;
- total iron concentration not to exceed 1,000 µg/L as a 30-day average;
- instantaneous dissolved iron concentration not to exceed 300 µg/L;
- instantaneous total manganese concentration not to exceed 1,000 µg/L; and
- instantaneous total aluminum concentration not to exceed 750 µg/L.

The designated use for Nanticoke Creek and other streams near the study area is CWF (Commonwealth of Pennsylvania, 2002). No published designated use is presently available for Newport Creek, but criteria for the CWF designation is useful for evaluating existing water-quality conditions.

Additional water-quality criteria established by the U.S. Environmental Protection Agency (1997) for the protection of freshwater aquatic life also have been adopted by the Commonwealth of Pennsylvania (2001b). These criteria include recommended continuous and maximum exposure limits for trace

metals and other constituents in water that could be present in AMD. Finally, criteria for protection of benthic aquatic organisms from metals in streambed or lakebed sediments are available (MacDonald and others, 2000), but these sediment-quality guidelines have not been adopted by regulatory authorities in the United States.

Impairment of water quality in the Newport and Nanticoke Creek Basins from AMD and flow alterations has been recognized by the Pennsylvania Department of Environmental Protection (PaDEP). TMDLs for Newport Creek are proposed for development during 2007-2009 (Pennsylvania Department of Environmental Protection, 2004a). Nanticoke Creek and Espy Run are on the Pennsylvania list of impaired waters, but no TMDL currently is planned (Pennsylvania Department of Environmental Protection, 2004b). Recently, attempts have been made to abate water losses in Nanticoke Creek (Ackman and Dilmore, 2002) and to passively treat contaminated water from an unnamed seep along Espy Run (Klemow, 2002b) and a portion of AMD from the Truesdale Mine (Klemow, 2003).

The general methods of passive treatment to remove iron, aluminum, and to a lesser extent manganese—in terms of matching treatment strategies with influent chemistry—are summarized in figure 7. Processes that are important for removal of dissolved metals from mine drainage include neutralization of acidity and the oxidation and hydrolysis of dissolved metals followed by settling of suspended precipitates

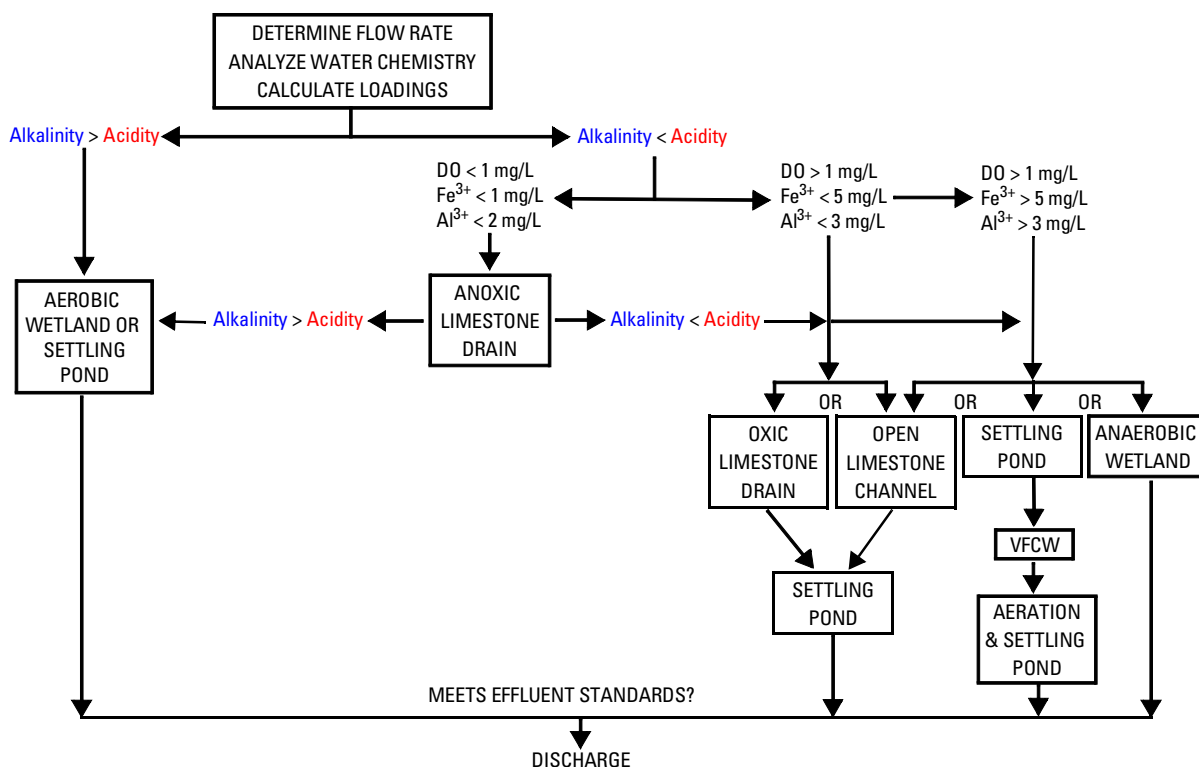


Figure 7. Flow chart for selection of passive treatment strategies (modified from Hedin and others (1994), Skousen and others (1998), and Pennsylvania Department of Environmental Protection (1999)). Vertical-flow compost wetland (VFCW) is also known as SAPS or RAPS.

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from the water column. Depending on the influent chemistry, a passive-treatment approach may facilitate metal removal by adding oxygen to oxidize reduced metals, increasing alkalinity to buffer protons produced by hydrolysis reactions, modifying retention time, or some combination of these.

In general, aerobic wetlands and ponds can treat net alkaline water; anoxic limestone drains (ALD) or oxic limestone drains (OLD) can treat acidic water with low concentrations of aluminum, Fe^{3+} , and DO; and vertical-flow compost wetlands (VFCW), anaerobic wetlands, and oxic limestone channels (OLC) can treat acidic water with high concentrations of aluminum, Fe^{3+} , and DO (Hedin and others, 1994; Skousen and others, 1998, 2000). These passive-treatment methods commonly are used in combination with aerobic wetlands. For example, mine water may initially be routed through relatively deep settling ponds that capture most of the suspended metals; alkalinity may then be increased with VFCW before routing the amended water into a downstream aerobic wetland where remaining metals can settle from the water column (fig. 7).

Aerobic wetlands or ponds commonly are used in the final step of most passive-treatment scenarios because they promote iron oxidation, hydrolysis, and deposition once pH and alkalinity are great enough to favor metal removal. Iron-removal rates per unit area of wetland, based on data from wetlands used to treat mine water draining from bituminous coal mines in western Pennsylvania, have been determined (Hedin and others, 1994). Generally, the rate of iron removal in wetlands increases as the pH and concentrations of iron and alkalinity increase (Kirby and others, 1999). Iron-removal rates as high as $20 \text{ (g/m}^2\text{)/d}$ (180 (lb/acre)/d) and as low as $5 \text{ (g/m}^2\text{)/d}$ (45 (lb/acre)/d) have been recommended for sizing aerobic ponds and wetlands to treat net alkaline and net acidic discharges, respectively (Hedin and others, 1994). As the rate of iron removal decreases, wetland size must increase to remove an equivalent amount of iron. For example, to remove the same amount of iron from the water column, a wetland sized for a removal rate of $5 \text{ (g/m}^2\text{)/d}$ would need be approximately four times larger than a wetland sized for a removal rate of $20 \text{ (g/m}^2\text{)/d}$.

Study Design and Methods

This study was designed to provide data for characterizing the effects of coal mining and AMD including (1) characterization of water quality before and after entering abandoned mines, (2) determination of variations in water quality under different flow conditions, and (3) identification of remedial priorities and strategies for remediation of AMD sources. Monitoring for the study was conducted by the USGS with assistance from the EC and WU. Assessment field work included (1) continuous stage measurement at 7 sites; (2) synoptic water-quality and flow sampling at 21 sites during June 2–4, 1999, and at 24 sites during October 7–8, 1999; and (3) measurements of flow and water

quality at 26 additional sites not included in the synoptic sampling effort.

The intent of synoptic sampling in June and October 1999 was to determine the variability of water quality during low and high base-flow conditions, respectively. However, continuous flow data computed on Newport Creek and South Branch upstream of their confluence (sites 39 and 48, respectively), Newport Creek downstream of South Branch (site 49), Nanticoke Creek and Espy Run above their confluence (sites 23 and 28, respectively), Nanticoke Creek at Warrior Gap (site 3), and at the outfall of the Susquehanna Number 7 Mine discharge (site 50; fig. 1, table 1) indicate the June and October measurements were similar to typical conditions (between the 25th and 75th percentile at gaged sites) rather than extremes measured during the study period (April 1999 through September 2000). Because remedial treatment systems commonly are designed for median or average flow conditions, these data may well serve to guide selection or design of treatment strategies. However, pH, DO, and metal concentrations can vary considerably with flow. Characterization of this variability was accomplished through sampling (by USGS and WU) over a range of flow conditions, including near maximum and minimum flows measured during this study. Data collected by USGS and WU, along with computations of acidity and metal loads, are presented in appendix 2.

Computation of Continuous Flow Record

Daily-mean flow was computed from continuous (15-minute interval) water-level (stage) measurements in the headwaters of Nanticoke Creek (site 3), downstream of the Truesdale Mine Discharge at Loomis Park (site 23), on Espy Run (site 28), near the confluence of South Branch and Newport Creek (sites 39, 48, and 49), and at the Susquehanna Number 7 Mine discharge (site 50) (fig. 1). Stage was recorded with Stevens type A71 graphical recorders from April 1999 through March 2000 [see fig. 8 for an example of a streamflow-gaging station (gage)]. Stilling wells, which minimize spurious water-level fluctuations, were installed at each gage except at the Susquehanna Number 7 Mine discharge. A weir was installed at each gage except Nanticoke Creek downstream of the Truesdale Mine Discharge (site 23) and the Susquehanna Number 7 Mine discharge (site 50) (fig 9). The weirs provided a more stable control than the natural stream channel, allowing for more accurate computation of flow. Continuous streamflow was computed from ratings that related the instantaneous flow measurements and corresponding stage (Rantz and others, 1982a, 1982b). Instantaneous flow was determined from streamflow velocity measurements made by wading across the channel with a vertical-axis current meter following standard methods described in Rantz and others (1982a).

To complement the flow data, a tipping-bucket rain gage was set up on the roof of the EC building near Nanticoke, Pa., for the measurement of precipitation during the study period; however, this equipment failed to produce useful results.

Table 1. Sampling sites in Newport and Nanticoke Creek Basins, Luzerne County, Pennsylvania.

[Gray shading indicates abandoned mine drainage is discharged at site (if flowing); mi², square miles; --, no data; Cr, Creek; yd yards; ft, feet; us, upstream; ds, downstream; AMD, abandoned mine drainage; CSO, combined sewer overflow; RR, railroad; SR, state route]

Site number	U.S. Geological Survey identifier	Scarlift number ¹	Drainage area (mi ²)	Site description	Latitude ²	Longitude ²	Altitude ³ (ft)	Synoptic sample	
								June 1999	October 1999
Nanticoke Creek Basin									
1	01537520	--	0.82	Nanticoke Cr headwaters at pond	41°10'51"	75°56'50"	980	Y	Y
2	01537522	--	.31	Unnamed tributary to Nanticoke Cr	41°10'49"	75°57'08"	930	Y	Y
3	01537524	N-8	1.32	Nanticoke Cr gage at Warrior Gap	41°10'59"	75°57'05"	860	Y	Y
4	0153752420	--	1.35	Nanticoke Cr 75 yd ds gage at Warrior Gap	41°11'02"	75°57'06"	815	N	N
5	0153752440	--	1.37	Nanticoke Cr 110 yd ds gage at Warrior Gap	41°11'04"	75°57'07"	800	N	N
6	0153752450	N-8A	1.41	Nanticoke Cr 260 yd ds gage at Warrior Gap	41°11'05"	75°57'08"	760	N	N
7	0153752460	--	1.48	Nanticoke Cr 380 yd ds gage at Warrior Gap	41°11'07"	75°57'12"	740	N	N
8	01537525	N-8B	1.58	Nanticoke Cr at Warrior Run	41°11'05"	75°57'33"	730	N	N
9	01537526	N-6	1.72	Nanticoke Cr us of Leuder Cr	41°11'22"	75°58'01"	615	N	N
10	01537527	--	.30	Leuder Cr headwaters	41°10'29"	75°58'24"	980	Y	Y
11	0153752750	N-9A	.58	Leuder Cr headwaters us RR Bed	41°10'42"	75°58'31"	900	N	N
12	01537528	--	.66	Leuder Cr headwaters at RR Bed	41°10'48"	75°58'38"	730	N	N
13	01537529	--	.24	Hanover Reservoir outflow	41°10'28"	75°59'21"	1,040	Y	Y
14	0153752920	N-13A	.29	Hanover Reservoir outflow us of pit	41°10'36"	75°59'21"	930	N	N
15	0153752940	--	.33	Hanover Reservoir outflow ds of pit	41°10'41"	75°59'21"	880	N	N
16	01537531	N-9	.91	Leuder Cr us of silt pond	41°11'11"	75°58'29"	650	N	N
17	0153753150	N-7	1.24	Leuder Cr ds of silt pond	41°11'17"	75°58'11"	635	N	N
18	01537532	--	1.26	Leuder Cr at Hanover road	41°11'19"	75°58'07"	610	N	Y
19	0153753210	--	.97	Leuder Cr at mouth	41°11'23"	75°58'04"	615	N	N
20	01537533	N-6	.03	Mine-pit outflow to Nanticoke Cr	41°11'28"	75°57'49"	595	N	N
21	01537534	--	3.77	Nanticoke Cr ds of Middle Road	41°11'55"	75°57'54"	580	N	N
22	411203075575901	N-4	--	Truesdale Mine Discharge (Dundee Outfall)	41°12'03"	75°57'59"	580	Y	Y
23	01537535	--	4.03	Nanticoke Cr gage ds Truesdale Mine Discharge	41°12'16"	75°58'17"	550	Y	Y
24	01537538	N-11	.74	Espy Run headwaters	41°11'20"	75°59'19"	590	N	N
25	01537540	N-12	.84	Unnamed tributary to Espy Run	41°11'21"	75°59'23"	590	N	N
26	01537541	N-10	2.19	Espy Run mainstem	41°11'40"	75°59'16"	590	N	N
27	01537542	N-10A	2.54	Espy Run mainstem	41°11'52"	75°59'11"	570	N	N
28	01537544	N-3A	2.94	Espy Run gage at Nanticoke, PA	41°12'17"	75°58'54"	530	Y	Y
29	01537545	N-1	7.51	Nanticoke Cr near mouth	41°12'32"	75°59'07"	510	Y	Y

Table 1. Sampling sites in Newport and Nanticoke Creek Basins, Luzerne County, Pennsylvania.—Continued

[Gray shading indicates abandoned mine drainage is discharged at site (if flowing); mi², square miles; --, no data; Cr, Creek; yd yards; ft, feet; us, upstream; ds, downstream; AMD, abandoned mine drainage; CSO, combined sewer overflow; RR, railroad; SR, state route]

Site number	U.S. Geological Survey identifier	Scarlift number ¹	Drainage area (mi ²)	Site description	Latitude ²	Longitude ²	Altitude ³ (ft)	Synoptic sample	
								June 1999	October 1999
Newport Creek Basin									
30	411036076041401	--	--	Newport Cr mine discharge at Glen Lyon	41°10'36"	76°04'14"	670	N	N
31	01537550	N8	1.46	Newport Cr headwaters at Glen Lyon	41°10'48"	76°03'38"	640	N	N
32	01537551	--	1.52	Newport Cr headwaters at Glen Lyon	41°10'56"	76°03'25"	640	N	N
33	01537553	N6	3.65	Newport Cr headwaters near Glen Lyon	41°11'08"	76°02'30"	590	N	N
34	01537555	--	.34	Newport Cr mainstem	41°11'36"	76°02'19"	570	N	N
35	411140076021501	--	--	Newport Cr mine seep us of mine pit	41°11'40"	76°02'15"	570	N	Y
36	01537556	--	5.87	Newport Cr us of mine pit	41°11'42"	76°02'09"	600	N	N
37	01537557	--	--	Newport Cr mine pit	41°11'39"	76°01'51"	600	Y	N
38	01537558	F2	--	Newport Cr mine-pit overflow	41°11'43"	76°01'50"	590	Y	Y
39	01537560	--	7.29	Newport Cr gage us of South Branch	41°12'08"	76°00'51"	550	Y	Y
40	01537561	--	1.65	South Branch at headwaters	41°10'34"	76°01'45"	630	N	N
41	411037076014201	--	--	South Branch discharge us of unnamed tributary	41°10'37"	76°01'41"	610	N	Y
42	01537563	M6	1.34	Unnamed tributary to South Branch	41°10'36"	76°01'40"	620	Y	Y
43	01537564	M5	3.49	South Branch downstream of unnamed tributary	41°10'49"	76°01'02"	590	Y	Y
44	01537566	M4	.52	Fairchild pond outlet	41°10'20"	76°00'27"	950	Y	Y
45	01537569	--	--	South Branch at Sheatown (AMD)	41°11'20"	76°00'45"	670	Y	Y
46	01537570	--	5.01	South Branch at Sheatown (CSO)	41°11'20"	76°00'45"	670	Y	Y
47	01537571	M1	5.56	South Branch at SR3001 Bridge	41°11'50"	76°00'45"	570	N	Y
48	01537572	--	5.87	South Branch gage near Mouth	41°12'07"	76°00'50"	550	Y	Y
49	01537574	--	13.2	Newport Cr gage ds of South Branch	41°12'13"	76°00'45"	560	Y	Y
50	411225076002201	--	--	Susquehanna Number 7 Mine discharge	41°12'25"	76°00'22"	570	Y	Y
51	01537578	N2	13.9	Newport Cr near mouth	41°12'44"	76°00'28"	510	Y	Y

¹Based on Scarlift reports for Newport Creek (Skelly and Loy, Inc., 1974) and Nanticoke Creek (GEO-Technical Services, Inc., 1975).

²Referenced to the North American Datum of 1983.

³Determined from the Nanticoke and Wilkes-Barre West 7.5-minute topographic quadrangles published by the U.S. Geological Survey. Vertical datum is National Geodetic Vertical Datum of 1929. Accuracy is ± 10 ft.



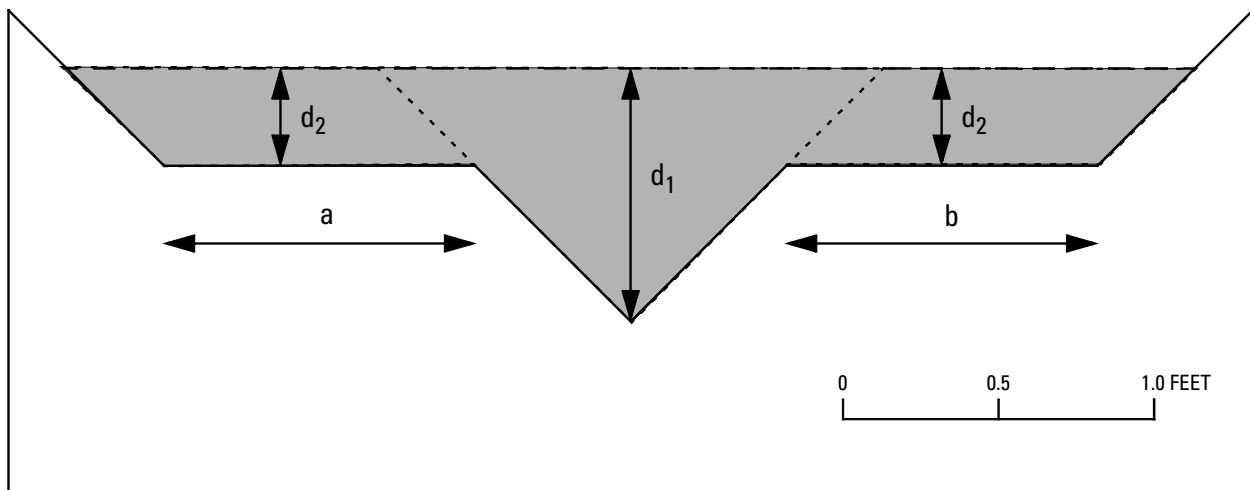
Figure 8. Stilling well and shelter housing instrumentation for gaging streamflow in Nanticoke Creek at Warrior Gap (site 3), Luzerne County, Pennsylvania.

Therefore, rainfall and air temperature data were obtained from a meteorological station at the Wilkes-Barre/Scranton International Airport approximately 20 mi northeast of the study area (Northeast Regional Climate Center, 2006). Although useful, these data were collected outside of the study area and may not entirely represent the local conditions experienced during this study. Thus, flow data computed at the streamflow-gaging stations described above were relied on for indicating drought and other hydrologic conditions.

Methods of Water-Quality Sampling and Analysis

In accordance with standard methods (Rantz and others, 1982a, 1982b; Wilde and others, 1998), field data for flow rate, temperature, SC, DO, pH, Eh, alkalinity, and acidity were measured using field-calibrated instruments when samples were collected. Water samples were collected manually as grab samples from well-mixed zones, avoiding bottom sediments and other debris, and then filtered (0.45- μm (micrometer) pore size) and preserved as appropriate for chemical or microbiological analysis (table 2).

The concentrations of major ions and dissolved metals in stream water and AMD samples were determined at the USGS National Water-Quality Laboratory in Denver, Colo., WU, or the PaDEP Laboratory in Harrisburg, Pa., by inductively coupled plasma atomic emission spectrometry (ICP-AES), ion chromatography (IC), colorimetry, or electrometric titration following methods in Greenberg and others (1992), Fishman and Friedman (1989), and Fishman (1993). Some stream-water samples also were analyzed for inorganic nitrogen (ammonia, nitrate, and nitrite) and phosphorus. Charge imbalances were calculated as the difference between cation and anion equivalents relative to the mean of cation and anion equivalents and were less than 10 percent for the synoptic samples in June and October 1999. Duplicates and USGS Standard Reference Water Samples that were submitted with each set of samples indicated precision and accuracy within 5 percent for all constituents. Deionized water blanks and filter blanks indicated concentrations below detection for all analytes.



Streamflow, Q (ft^3/s) = $Q_1 + Q_2$, where:

$$Q_1 = 2.47d_1^{2.5}$$

$$Q_2 = C(a+b)d_2^{1.5}$$

C is a coefficient ranging from 2.9 to 3.3

Figure 9. Conceptual diagram of weir used in this study and associated equations for computation of streamflow on the basis of depth of water flowing over weir plate (shaded area). Units for a , b , d_1 , and d_2 are in feet. See Rantz and others (1982a, p. 12–22; 1982b, p. 294–305) for detailed information on the use of weirs for measurement of discharge.

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Table 2. Analytical methods used to determine physical and chemical characteristics of water within Newport and Nanticoke Creek Basins.

[ft³/s, cubic feet per second; ft, feet; °C, degrees Celsius; mV, millivolts; μS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligrams per liter; μg/L, micrograms per liter; --, not applicable or no data; RU, raw, unpreserved; FU, filtered, unpreserved; FAS, filtered, acidified with H₂SO₄; FA, filtered, acidified with HNO₃; RA, raw, acidified]

Storet code	Description of characteristic or constituent	Analytical method number and type ¹		Detection limit	Sample prep ²
FIELD CHARACTERISTICS OR METHODS					
72020	Elevation water table, head (ft)	--	--	0.1	--
00061	Discharge (ft ³ /s)	--	--	0	--
00010	Temperature, water (°C)	--	--	.5	RU
00090	Oxidation reduction potential (Eh field) (mV)	--	E	10	RU
00095	Specific conductance (μS/cm at 25°C)	I-1780-85	E	10	RU
00300	Dissolved oxygen (mg/L)	--	E	5	RU
00400	pH, field (standard units)	--	E	1	RU
00410	Alkalinity field, endpoint pH 4.5 (mg/L CaCO ₃)	I-1030-85	ET	1	RU/FU
00435	Acidity, total, endpoint pH 8.3 (mg/L CaCO ₃)	³ I-1020-85	ET	5	RU/FU
LABORATORY CHARACTERISTICS OR METHODS					
90095	Specific conductance, lab (μS/cm at 25°C)	I-1780-85	E	10	RU
00403	pH, lab (standard units)	--	E	.1	RU
00915	Calcium, dissolved (mg/L)	I-1472-85	ICP	.02	FA
00925	Magnesium, dissolved (mg/L)	I-1472-85	ICP	.01	FA
00930	Sodium, dissolved (mg/L)	I-1472-85	ICP	.2	FA
00940	Chloride, dissolved (mg/L)	I-2057-85	IC	.1	FU
00945	Sulfate, dissolved (mg/L as SO ₄)	I-2057-85	IC	.1	FU
00955	Silica, dissolved (mg/L)	I-1472-85	ICP	.01	FA
01046	Iron, dissolved (μg/L)	I-1472-85	ICP	3	FA
01047	Iron, ferrous dissolved (μg/L)	315-B	PC	10	RA
01056	Manganese, dissolved (μg/L)	I-1472-85	ICP	3	FA
01106	Aluminum, dissolved (μg/L)	I-1054-85	AES	10	FA
00608	Nitrogen, ammonia (mg/L)	I-2522-90	CSH	.02	FAS
00631	Nitrogen, nitrate + nitrite (mg/L)	I-2545-90	CCR	.05	FAS
00613	Nitrogen, nitrite (mg/L)	I-2540-90	CCR	.01	FAS
00671	Phosphorus, orthophosphate (mg/L)	I-2601-90	CPM	.01	FAS

¹Field methods are described by Rantz and others (1982a, 1982b), Wood (1976), and Wilde and others (1998) Laboratory methods beginning with the letter I are described by Fishman and Friedman (1989) and Fishman (1993). The method for measurement of ferrous iron is described in Greenberg and others (1992). Acidity titration conducted using hydrogen peroxide without heating, following methods of Ott (1986).

Analytical methods by abbreviation: E, electrometric; ET, electrometric titration; IC, ion chromatography; AES, atomic emission spectrometry; ICP, inductively coupled plasma spectrometry; PC, phenanthroline colorimetry; CCR, colorimetry, cadmium reduction; CSH, colorimetry, salicylate-hypochlorite; CPM, colorimetry, phosphomolybdate.

²Sample preparation is indicated by abbreviations: Filtration was through nitrocellulose filters with 0.45-μm pores. Reagent-grade nitric acid was used to acidify all samples, except ferrous-iron samples, which were acidified with hydrochloric acid.

³Sample not heated.

During the initial study-area-wide synoptic sampling event on June 2-4, 1999, anecdotal evidence of sewage contamination, such as sewer odors and debris, was observed, especially in the lower reaches of Nanticoke Creek and South Branch. Because this synoptic sampling effort was geared toward inorganic constituents in AMD, bacteria and nutrients (nitrogen and phosphorus) commonly associated with sewage were not characterized. In a subsequent sampling event on June 24, 1999, to determine the extent of sewage contamination, samples for determination of *Escherichia coli* (*E. coli*) and total coliform bacteria and nutrient concentrations were collected upstream of the Truesdale Mine Discharge (site 21), at the mouth of Nanticoke Creek (site 29), Newport Creek upstream of South Branch (site 39), South Branch at Sheatown (site 46), South Branch upstream of Newport Creek (site 48), and on Newport Creek downstream of South Branch (site 49). These are all surface-water locations except for site 46, where water discharging from a concrete pipe was suspected to be a sewer overflow point.

Samples collected on June 24, 1999, were analyzed simultaneously for total coliform and *E. coli* bacteria using standard membrane-filtration techniques and the m-ColiBlue24 Broth nutritive medium (Hach Company, 2003). Commercially sterilized and individually wrapped gridded filters with 0.45- μm pores, media, and petri dishes were used. Samples were collected using glass bottles and stainless-steel filtration apparatus that had been sterilized by autoclaving. A 100-mL (milliliter) sample was vacuum filtered, and the filter immediately was placed on the nutritive media in a disposable petri dish, covered, inverted, and stored at 4°C for up to 5 hours. After collection, the inverted samples were incubated at 35°C for 24 hours in a portable incubator. Red colonies were counted as total coliform and blue colonies were counted as *E. coli*.

Loading of dissolved iron, manganese, and aluminum was computed from samples collected during the study. The metals loading is expressed in pounds per day and was computed as the product of flow rate and metal concentrations by:

$$\text{Metals Load (lb/d)} = 5.395Q \times [(C_{\text{Fe}} + C_{\text{Mn}} + C_{\text{Al}})/1,000] \quad (7)$$

where

Q is the instantaneous flow rate, in cubic feet per second, and C with subscript Fe, Mn, or Al is the iron, manganese, or aluminum concentration, respectively, in micrograms per liter.

Although acidity was measured on refrigerated, tightly sealed samples by cold peroxide titration, the interpretation of these measured acidities for AMD can be debated (Kirby, 2002; Kirby and Cravotta, 2004; Cravotta and Kirby, 2004). Therefore, acidity for all the samples was computed from pH and dissolved metals concentrations as:

$$\begin{aligned} \text{Acidity}_{\text{computed}} \text{ (mg/L CaCO}_3\text{)} &= 50 \times (10^{(3-\text{pH})}) \\ &+ 50 \times (0.002 \times C_{\text{Fe}}/55.85 + 0.002 \times C_{\text{Mn}}/54.94 \\ &+ 0.003 \times C_{\text{Al}}/26.98), \end{aligned} \quad (8)$$

where

C with subscript Fe, Mn, or Al is the dissolved concentration of the metal indicated by the subscripts Fe, Mn, and Al, in micrograms per liter.

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

$$\begin{aligned} \text{Net alkalinity (mg/L CaCO}_3\text{)} &= \\ &\text{Alkalinity} - \text{Acidity}_{\text{computed}} \end{aligned} \quad (9)$$

Note that net acidity ($\text{Acidity}_{\text{computed}} - \text{Alkalinity}$) has the same magnitude but is opposite in sign from net alkalinity. The net alkalinity on the basis of computed acidity (eqs. 8 and 9) and measured alkalinity should be similar 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, 2004).

For the purpose of identifying differences in water chemistry between synoptic samples collected in June and October 1999, a nonparametric, two-sided Wilcoxon signed-rank test was used (Helsel and Hirsch, 2002, p. 142). Sites were divided into two categories to evaluate differences in ground-water (AMD) samples and surface-water samples. Flow rate, metal loads, and 10 water-quality constituents were compared within each category. The null hypothesis for the Wilcoxon signed-rank test is there is no difference between median values in June and October. For this study, the null hypothesis was rejected if the p-value (probability that a difference occurs by chance) is less than 0.05.

The physical setting (area available for treatment, slope), maximum measured flow, and specific water-quality data for each AMD source including minimum net alkalinity (eq. 9) and maximum concentrations of DO and metals for the samples were used to identify potential passive treatment and other remediation strategies. 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 strategies (fig. 7). Consideration was given to various combinations of passive- and active-treatment strategies. The wetland

size computed using the 180 (lb/acre)/d (20 (g/m²)/d) iron-loading rate of Hedin and others (1994) was compared with available land area to indicate feasibility for implementation of passive treatment at each site.

The data assembled for this study have been incorporated into digital databases, including the USGS National Water Information System (NWIS), spreadsheets, and a Geographic Information System (GIS). Water-quality and instantaneous flow data collected by USGS and WU are summarized by site in appendix 2 and are accessible on the World Wide Web (U.S. Geological Survey, 2007).

Effects of Historical Coal Mining and Drainage from Abandoned Mines

The hydrologic characteristics of the Newport/Nanticoke Creek Basins have been profoundly altered by abandoned underground and surface-mining operations that enhance infiltration of surface runoff and stream water and facilitate interbasin transfer of ground water. Headwater streams draining northward from the uplands are unaffected by mining in the valley, but once Nanticoke Creek, South Branch, and their tributaries cross the area underlain by coalbeds, the streams lose water to the underground mine workings through fractures, crop falls, and subsidence features. Water that enters the underground mines, along with runoff from mine refuse, can readily dissolve FeS₂ oxidation products and convey SO₄²⁻, Fe²⁺, Mn²⁺, and other dissolved metals to AMD sites and nearby streams. DO concentrations may be depressed or depleted. If anoxic conditions prevail, Fe²⁺ can persist in solution rather than oxidizing to Fe³⁺ (eq. 3) and hydrolyzing to form Fe(OH)₃ (s) (eq. 4). When exposed to DO, the Fe²⁺ oxidation process can be slow, facilitating transport of the dissolved metal but complicating AMD treatment. For many approaches to treatment of AMD, the neutralization of acidity, the oxidation of Fe²⁺, and the settling of Fe(OH)₃ (s) from the water column are primary objec-

tives. This section describes the range and variability of flow and chemical constituents that are important for development of an appropriate remedial strategy.

Hydrology

Hydrologic conditions from April 1999 through March 2000, the period of continuous flow measurement and most water-quality-data collection, were variable, especially at sites with appreciable surface-water influence. At the Wilkes-Barre/Scranton weather station, rainfall totals in June, July, and August 1999 were 23, 56, and 38 percent below their respective normals (Northeast Regional Climate Center, 2006), resulting in steadily decreasing flows that reached their minima in August 1999. Just as the study area and other basins to the south were experiencing drought, remnants of Hurricane Floyd brought 5.7 in. of rain during September 14 to 16, 1999. This event and additional rain contributed 9.8 in. of cumulative precipitation in September, compared to a 30-year mean of only 3.6 in. (Northeast Regional Climate Center, 2006; fig. 10). Runoff following the September rains contributed to the highest measured flows during the study. Water that did not run off into streams infiltrated and recharged ground water.

Generally, upland streams that had extensive intermittent segments or that had only small AMD contributions, such as Nanticoke Creek at Warrior Gap, Espy Run, South Branch, and Newport Creek downstream of South Branch, had greater flow variability than large AMD sources and associated downstream reaches, as indicated by the coefficient of variation (table 3). The coefficient of variation for flows at these upland sites ranged from 1.56 to 2.22, compared to values of only 0.20 to 1.13 for the large AMD sites. The large AMD sources and downstream sites had smaller coefficients of variation because the precipitation that recharged the mines was released slowly from storage as base flow instead of rapidly entering the stream as runoff. For example, flow from the Susquehanna Number 7 Mine (site 50), the largest discharge that enters near the mouth

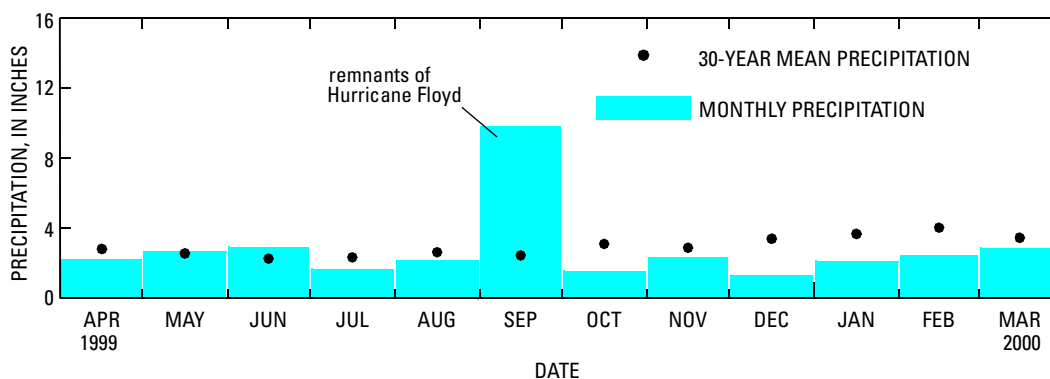


Figure 10. Precipitation at Wilkes-Barre/Scranton International Airport, located approximately 20 miles northeast of Newport and Nanticoke Creek Basins, Luzerne County, Pennsylvania. Data are from the Northeast Regional Climate Center (2006).

Table 3. Summary statistics for daily mean flow from gaged stream and mine-discharge sites in Newport and Nanticoke Creek Basins, April 1999 - March 2000, Luzerne County, Pennsylvania.

[Gray shading indicates abandoned mine drainage is discharged at site; ds, downstream; us, upstream; N, number of values; Min, minimum; P25, 25th percentile; P50, 50th percentile or median; P75, 75th percentile; Max, maximum; coefficient of variation is standard deviation divided by mean]

Site number	Description	N	Daily mean flow rate, in cubic feet per second						Standard deviation	Coefficient of variation
			Min	P25	P50	P75	Max	Mean		
3	Nanticoke Creek at Warrior Run	345	0.079	0.3	1.0	2.2	28	1.8	2.8	1.56
23	¹ Nanticoke Creek ds Truesdale Mine Discharge	327	.019	1.2	4.5	7.9	40	6.3	7.1	1.13
28	Espy Run	344	.262	.5	1.2	5.8	74	8.3	15.5	1.87
39	Newport Creek us South Branch	223	.175	.7	2.0	3.9	8.2	2.2	1.6	.73
48	South Branch	321	.041	.3	.8	2.0	65	2.3	5.1	2.22
49	Newport Creek ds South Branch	275	.428	1.6	3.0	5.3	71	5.6	9.1	1.63
50	Susquehanna Number 7 Mine discharge	293	4.9	5.9	7.0	8.1	11	7.1	1.4	.20

¹Truesdale Mine Discharge is locally referred to as the Dundee Outfall.

of Newport Creek, had the smallest coefficient of variation (only 0.20). Conversely, coefficients of variation for flows in South Branch (site 48) and Espy Run (site 28) were large (2.22 and 1.87; table 3) because these streams receive no appreciable volumes of AMD to moderate flow variability. Other sites had flows with intermediate coefficients of variation, including site 3, which is in the unmined headwaters of Nanticoke Creek. Variability of flow at site 3 may be moderated by an upstream lake near Witinski Villa (fig. 1), causing the coefficient of variation to be low compared to other upland sites.

Streamflow losses between site 3 in the headwaters of Nanticoke Creek and site 22 at the Truesdale Mine Discharge caused this intermediate reach of Nanticoke Creek to be dry during most of the study except for short periods (days) following major rainfall events. Contributions of AMD from site 22, an ungaged location, were the predominant source of streamflow at the gaging station on Nanticoke Creek at site 23, about 0.5 mi downstream (fig. 11). However, after the Truesdale Mine temporarily ceased discharging on about June 29, 1999, streamflow at site 23 decreased by 98 percent over the following 2-week period (from 1.7 to 0.03 ft³/s; fig. 11). Recharge by surface-water losses, precipitation, and ground-water infiltration to the mine pool were not great enough to maintain the discharge from the Truesdale Mine; the mine-pool elevation dropped below 573.2 ft mean sea level, the point of zero flow (GEO-Technical Services, Inc., 1975; Walski and others, 1997). Even when the Truesdale Mine Discharge and the upper reaches of Nanticoke Creek were dry, streamflow at site 23 was maintained at a very low rate.

In Newport Creek, discharge from the Susquehanna Number 7 Mine (site 50) entering approximately 0.5 mi downstream of South Branch contributed the largest volume of flow; the median discharge was 7.0 ft³/s from April 1999 to March 2000 (table 3; fig. 12). During the relatively dry months of June, July, and August 1999, flow from the Susquehanna Number 7 Mine steadily decreased but was still greater than combined flows measured at the upstream gages on Newport Creek (site 39) and

South Branch (site 48) (fig. 12). Flow data collected during synoptic sampling events also indicated additional inflows were present upstream and downstream of site 50 and may contribute unidentified AMD to Newport Creek.

Flows from the unmined subbasin of Nanticoke Creek (area draining to site 3) were used to estimate flow at other sites that are influenced by mining, allowing an evaluation of departure from an unmined reference condition. Flows were estimated by multiplying the yield (ft³/s)/mi² at site 3 for June and October 1999 by the contributing drainage area for each of the other sites (table 1, fig. 13). Note that drainage area as used here refers to the surface drainage area to the site where flow was measured and does not include underground areas such as mines that extend underneath surface-drainage divides. Some flows measured in headwaters (like at sites 1 and 13) and locations downstream of AMD sources (like at sites 51 and 29), correlate well with flows estimated from site 3, plotting near the line of equality in figure 13. Many sites where stream water is lost to the underground mines (sites 42 and 46 for example) plot far below the line of equality in figure 13 because flows were much less than estimated.

Differences in the relation between the measured and estimated flows are substantial at most stream sites overlying mined coal seams. Such differences indicate (1) streamflow was lost to the underground mines in intermediate reaches, (2) ground-water flow to discharge points could cross surface-drainage divides, and (3) discharges of AMD in lower reaches restored volumes lost in upstream reaches. These factors vary within each subbasin and with weather conditions. For example, the flow at the mouth of Nanticoke Basin (site 29) was 3.8 ft³/s in June 1999 and 8.2 in October 1999, but flows at the mouth of Newport Creek (site 51) were substantially greater in June (16 ft³/s) than in October (9.7 ft³/s) (figs. 14 and 15). Varying recharge rates and amounts of interbasin transfer of ground water between Newport and Nanticoke Basins may explain these differences.

18 Effects of Historical Coal Mining and Drainage from Abandoned Mines in Newport and Nanticoke Creeks, 1999–2000

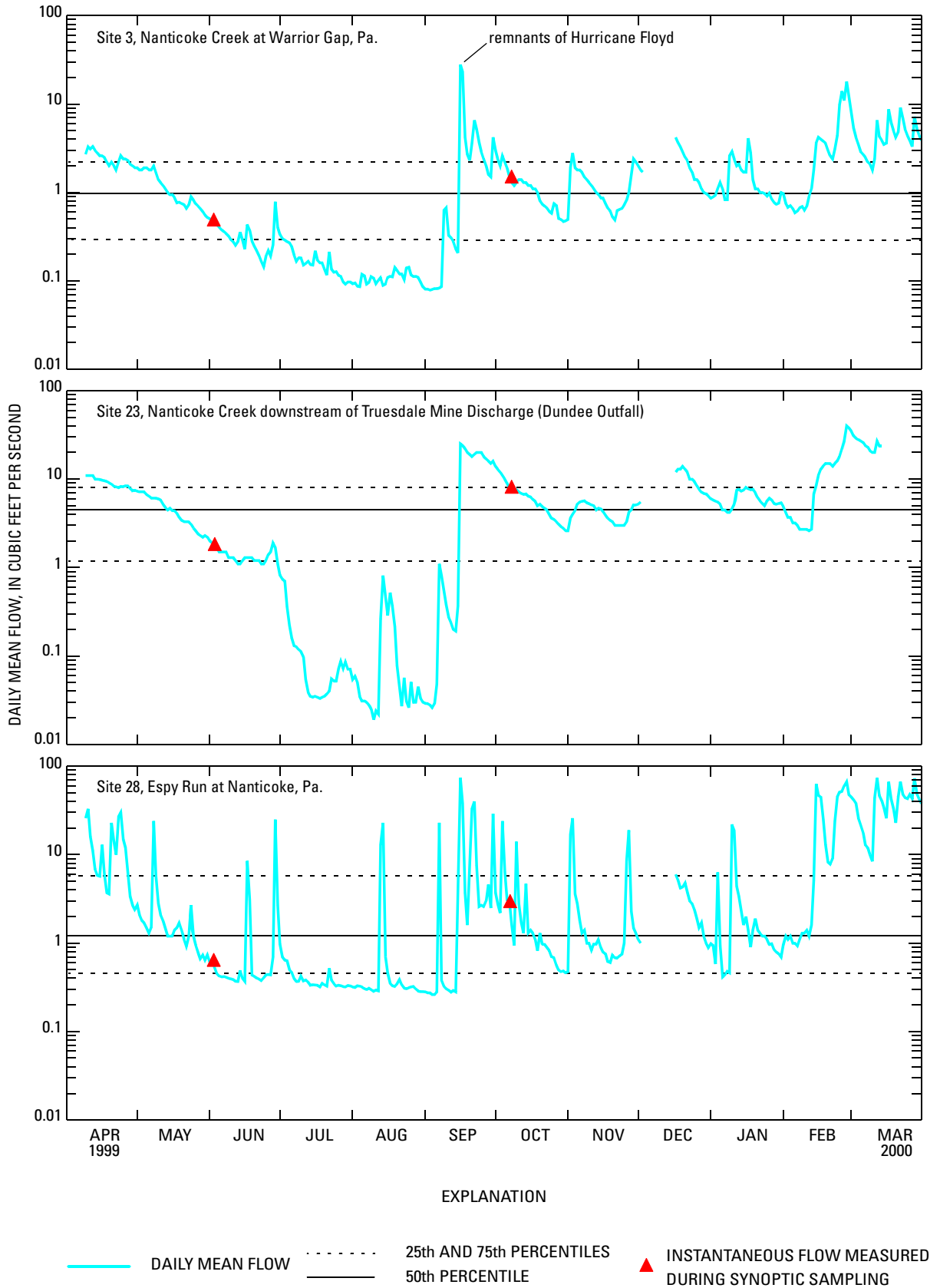


Figure 11. Flow hydrographs for selected sites in the Nanticoke Creek Basin, Luzerne County, Pennsylvania.

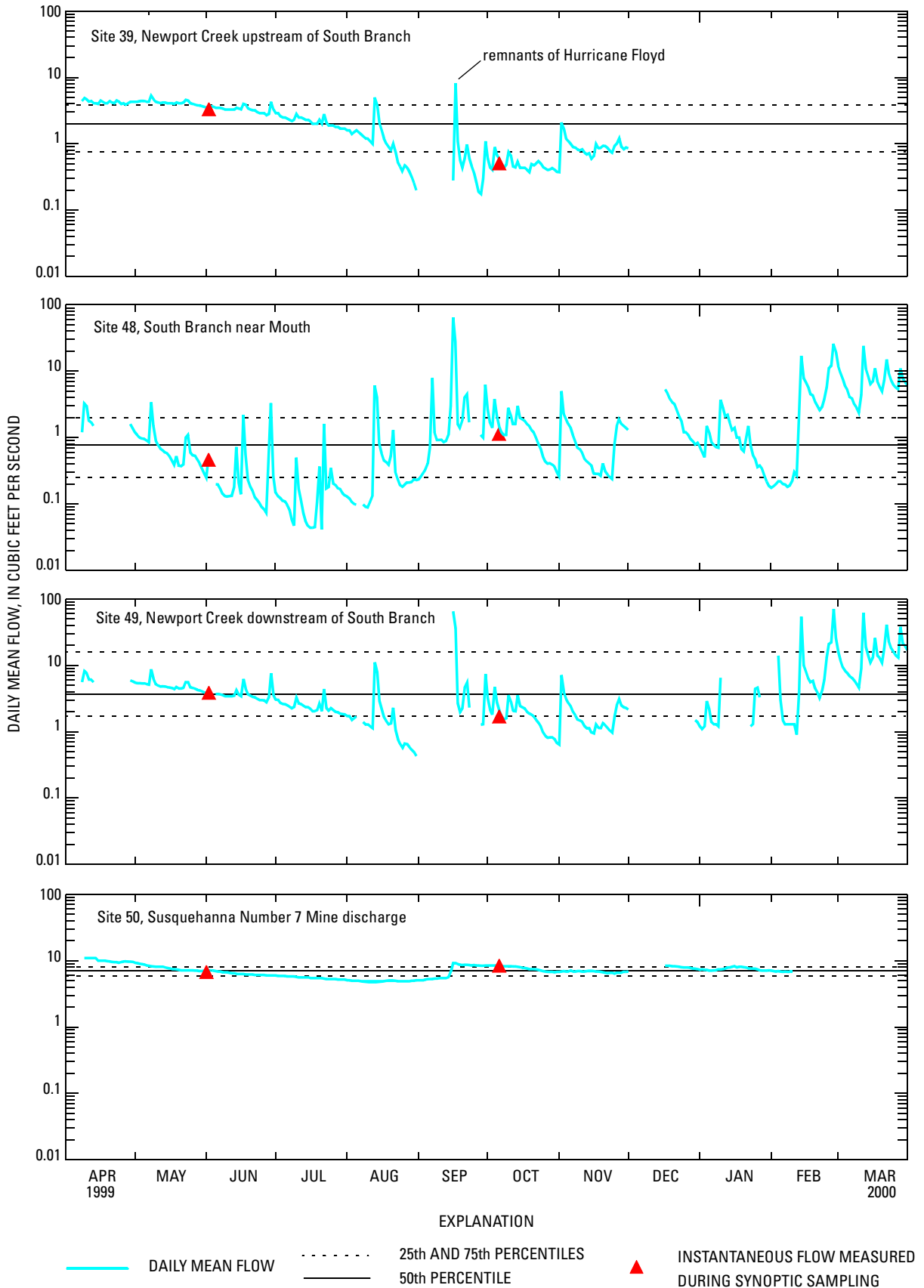
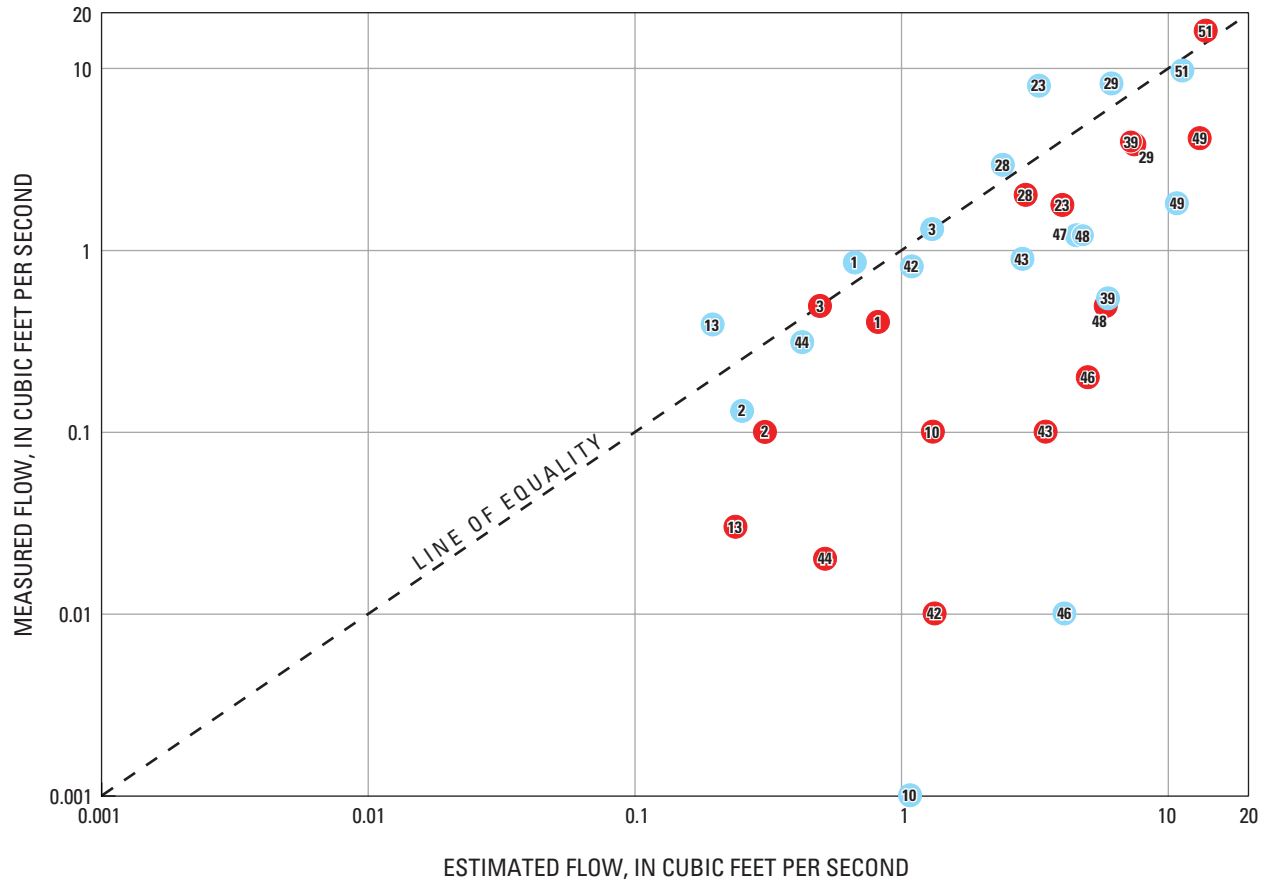


Figure 12. Flow hydrographs for selected sites in the Newport Creek Basin, Luzerne County, Pennsylvania.



EXPLANATION

- ④ FLOW DETERMINED JUNE 4-6, 1999
- ④ FLOW DETERMINED OCTOBER 7-8, 1999

Figure 13. Relation between estimated and measured flows at stream sites in Newport and Nanticoke Creek Basins, Luzerne County, Pennsylvania. Estimated flows are computed from yields (cubic feet per second per square mile) at site 3, a 1.32-square mile, unmined subbasin in the headwaters of Nanticoke Creek. Site numbers are listed in table 1.

Stream-Water and Mine-Discharge Quality

Results of Wilcoxon signed-rank tests indicate the chemical properties of AMD (ground water) were not significantly different between June and October 1999 (table 4). Nevertheless, characteristics like temperature, DO, chloride, and aluminum for surface water sampled in June and October were significantly different (p-values less than 0.05). At the surface-water sites, water temperature was cooler (p-value was 0.0005; fig. 16B) and DO concentration was greater (p-value 0.0067; fig. 16C) in October than in June because cooler water has greater capacity to contain DO and other gases than warmer water (Hem, 1985). Concentrations of dissolved aluminum were higher in October than in June (p-value was 0.0202,

fig. 16F) because of aluminum contributed to the headwaters of South Branch from a small AMD discharge at site 41 (concentration in October 1999 was 6,700 µg/L) that was not sampled and probably was not flowing in June 1999. Even though some AMD discharges were flowing during one sampling event but not the other, ground- or surface-water flows were not statistically different (table 4). Values for SC and chloride generally were smaller in October than in June 1999 (figs 16D and 16E; table 5), possibly because of freshening of recharge after two seasons without road-salt additions.

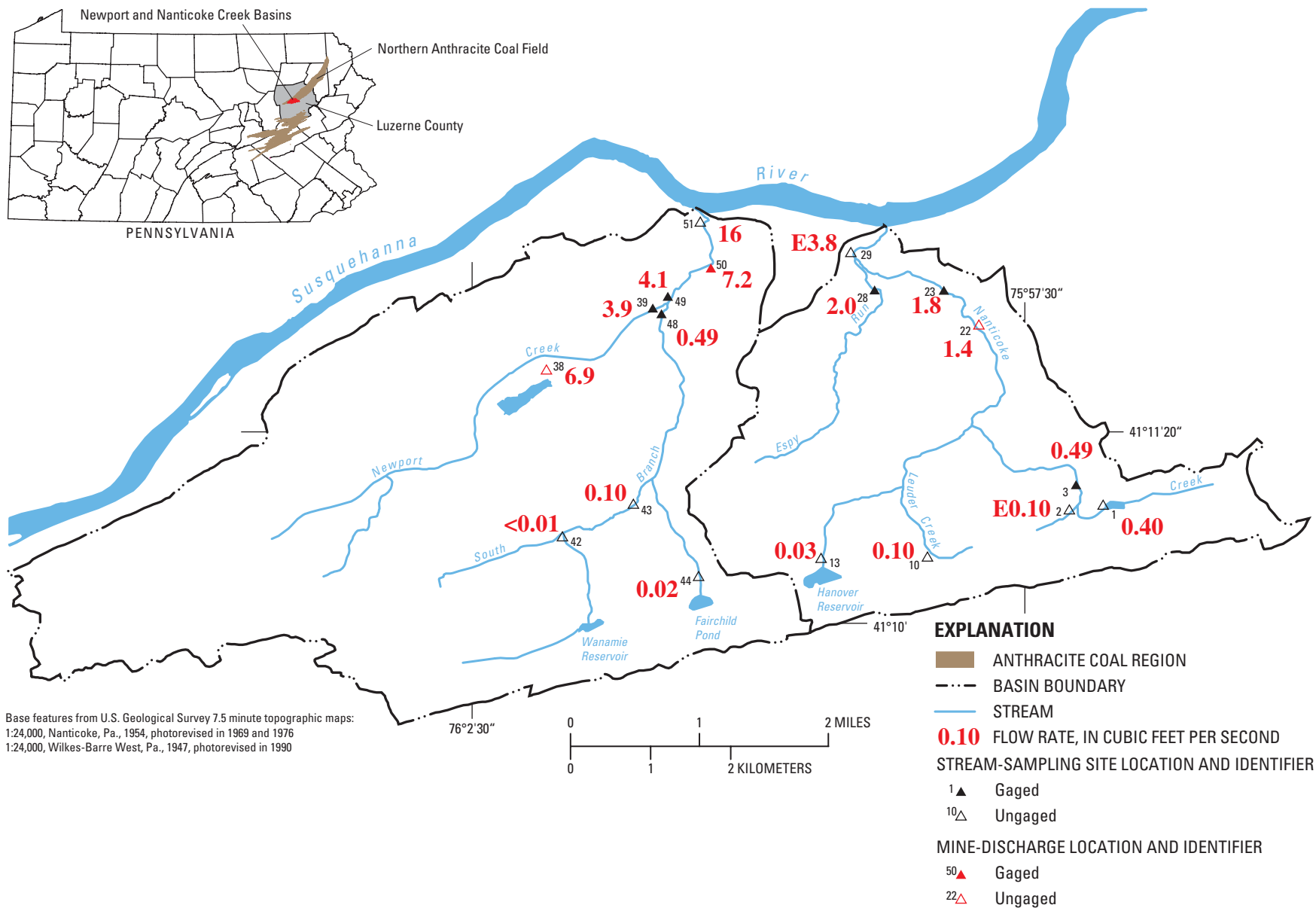


Figure 14. Spatial variability of flow at selected locations throughout Newport and Nanticoke Creek Basins, June 2-4, 1999, Luzerne County, Pennsylvania (E, estimated; <, less than).

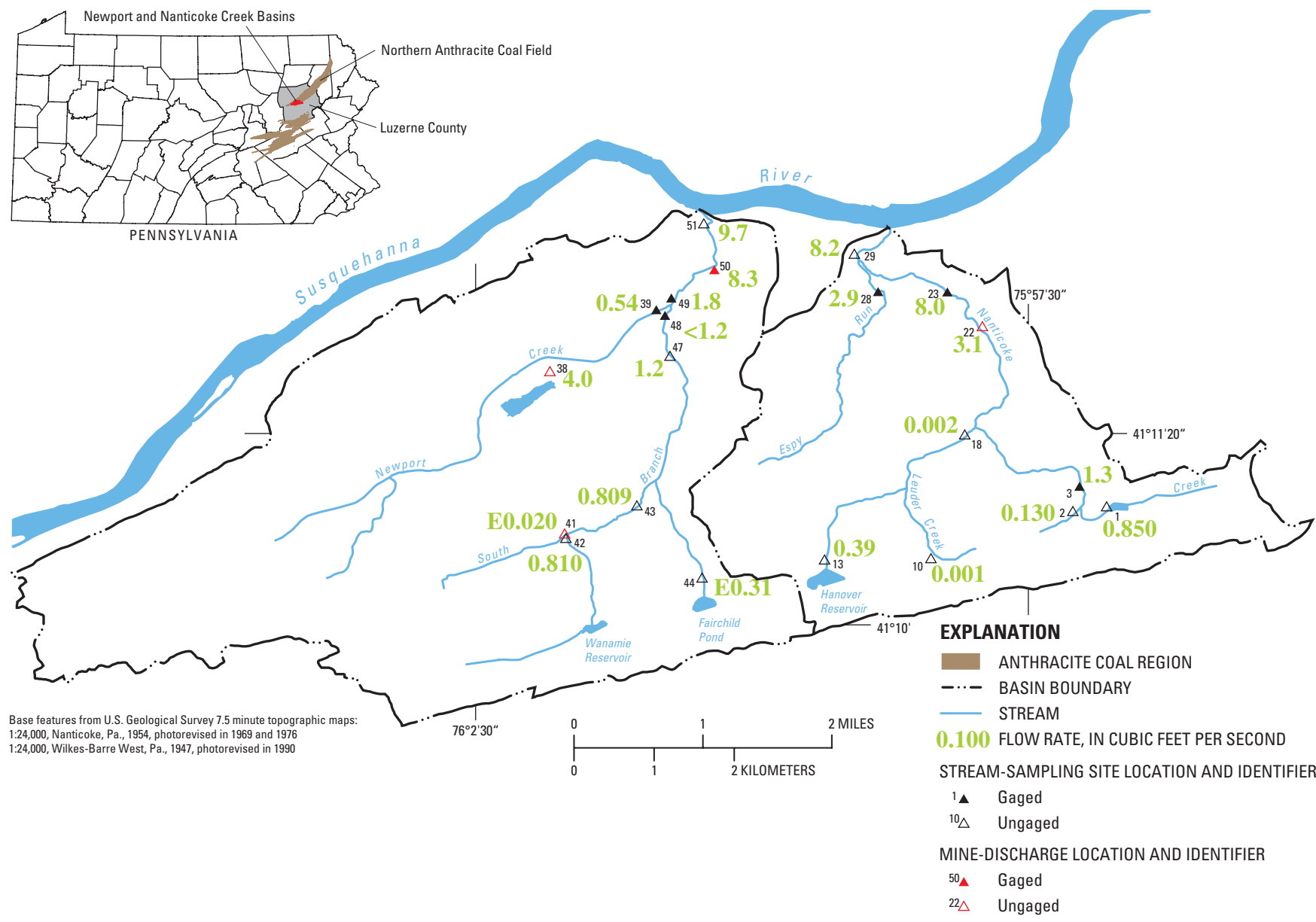


Figure 15. Spatial variability of flow at selected locations throughout Newport and Nanticoke Creek Basins, October 7-8, 1999, Luzerne County, Pennsylvania (E, estimated; <, less than).

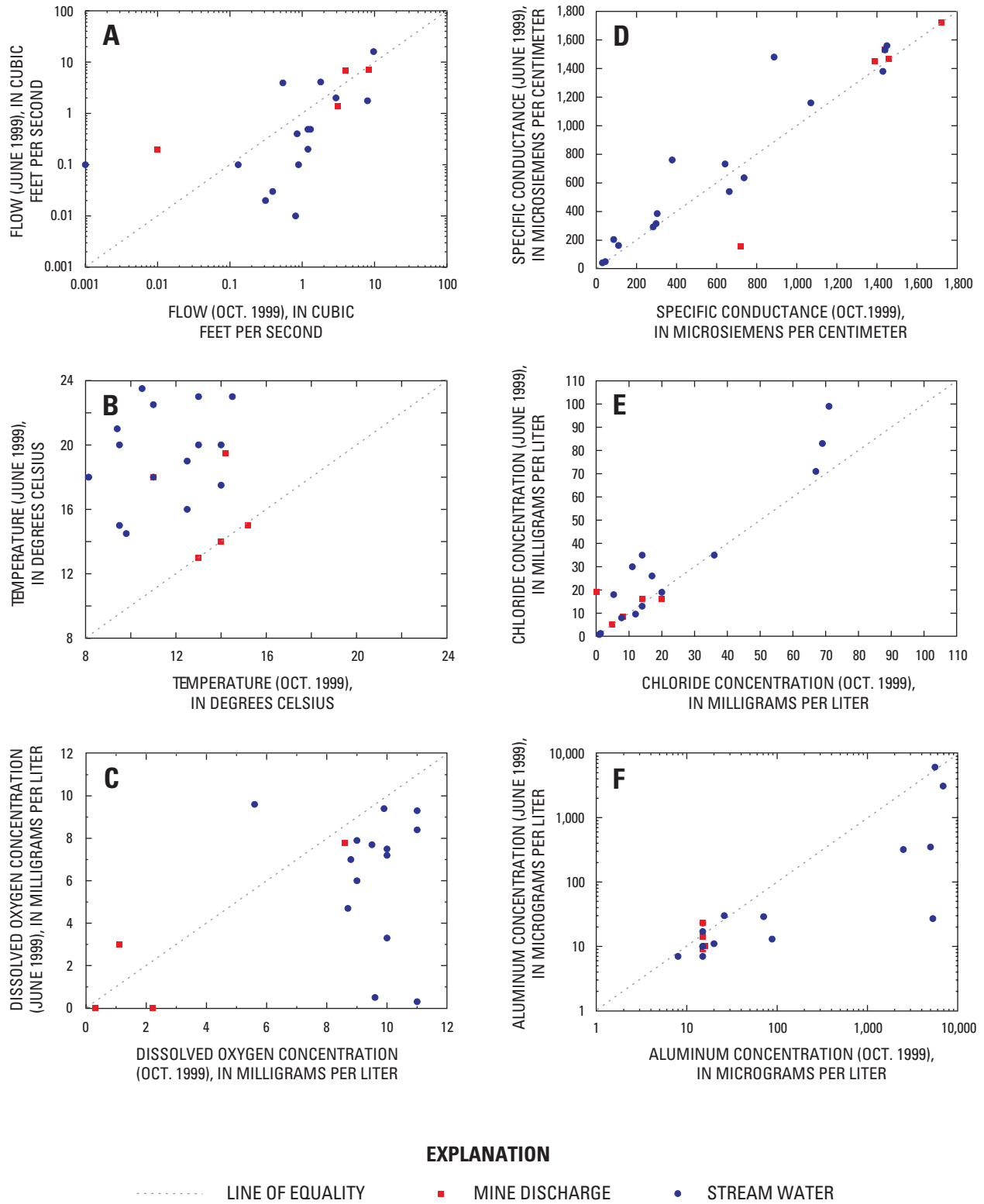


Figure 16. Comparison of June and October 1999 samples for flow (A) and selected water-quality constituents (B-F), Newport and Nanticoke Creek Basins, Luzerne County, Pennsylvania.

Table 4. Statistical results from comparison of flow and water quality between June and October 1999 sampling events, Newport and Nanticoke Creek Basins, Luzerne County, Pennsylvania.

[N, number of paired samples; ft³/s, cubic feet per second; μ S/cm, microsiemens per centimeter at 25 degrees Celsius; °C, degrees Celsius; mg/L, milligrams per liter; lb/d, pounds per day; Fe, dissolved iron; Mn, dissolved manganese; Al, dissolved aluminum; μ g/L micrograms per liter; shading indicates a significant difference]

Constituent	Ground water only		Surface water only	
	p-value ¹	N	p-value ¹	N
Flow rate (ft ³ /s)	1.0000	4	0.3303	15
Temperature (°C)	.4795	5	.0005	16
Dissolved oxygen (mg/L)	.6250	4	.0067	14
pH	.7865	5	.2423	16
Specific conductance (μ S/cm)	.6831	5	.0525	16
Net alkalinity (mg/L CaCO ₃) ²	1.0000	4	.0931	13
Chloride (mg/L)	.4375	5	.0838	14
Sulfate (mg/L)	.5879	5	.6153	14
Aluminum, dissolved (μ g/L)	.8539	4	.0202	14
Iron, dissolved (μ g/L)	.7103	4	.5927	14
Manganese, dissolved (μ g/L)	1.0000	4	.6832	14
Fe+Mn+Al load (lb/d)	.6250	4	.3460	14

¹For this study, the null hypothesis was rejected if the p-value (probability that a difference occurs by chance) is less than 0.05

²Net alkalinity = measured alkalinity - computed acidity.

Water-Quality Variations

The synoptic data for June and October 1999 (table 5), along with additional data collected intermittently during 1999–2000 (see appendix 2), indicate headwaters sampled upstream of the mined area (sites 1, 2, 3, 13, and 44) generally were oxygenated (DO concentration range was 4.7 to 11.0 mg/L), near-neutral (pH range was 5.8 to 7.6), and net alkaline (net alkalinity range was 2.0 to 25.0 mg/L CaCO₃), with relatively low concentrations of sulfate (6.40 to 24.0 mg/L) and metals (less than 500 μ g/L of dissolved iron, manganese, and aluminum). SC was lower at sites 14 and 44 in the headwaters of Newport Creek (range was 30 to 163 μ S/cm) than at sites 1, 2, and 3 in Nanticoke Creek headwaters (range was 185 to 400 μ S/cm). Because sites 1, 2, and 3 are upstream of the mined area, it is unlikely that dissolved sulfate and metals associated with AMD were causing the elevated SC. Instead, SC was probably elevated because of chloride in these samples that could be derived from road-deicing salts applied during the winter months. Higher concentrations of chloride were measured under base-flow conditions in June than in October 1999, indicating chloride enters the shallow ground water and streams in the winter months but is gradually flushed from the system during the spring and summer.

Although the synoptic data are useful for describing spatial variation at points in time, more frequently collected water-

quality samples and corresponding flows better described temporal variations. Continuous discharge and additional water-quality data indicate the most pronounced changes in water chemistry were in the lower reaches of South Branch and in Nanticoke Creek downstream of the Truesdale Mine Discharge (site 23). Some mechanisms influencing the water chemistry in each basin are the same, such as oxidation of pyrite and the hydrolysis of iron and other metals. Other factors, such as sewage contamination, may play a greater role in South Branch than in the Nanticoke Basin.

Bacteria and nutrient data, along with corresponding concentrations of ammonia and nitrite plus nitrate, indicate sewage contamination was prevalent in the lower reaches of Nanticoke Creek, South Branch, and Newport Creek below South Branch. Samples collected at sites 21, 29, 39, 46, 48, and 49 on June 24 and October 7–8, 1999, had positive counts for total coliform and *E. coli* bacteria along with other sewage- and AMD-related contaminants (see appendix 2). *E. coli* concentrations ranged from 4 col/100 mL in Newport Creek downstream of South Branch (site 49) to >200 col/100 mL in Nanticoke Creek upstream of the Truesdale Mine Discharge (site 21) and were 120 col/100 mL or greater at the mouths of Nanticoke Creek (sites 29) and South Branch (site 48). Concentrations of ammonia ranged from 0.12 to 1.3 mg/L and generally were greater than concentrations of nitrite plus nitrate.

Table 5. Values of selected constituents in stream water and abandoned mine drainage sampled during June and October 1999, Newport and Nanticoke Creek Basins, Luzerne County, Pennsylvania.

[Gray shading indicates abandoned mine drainage is discharged at site (if flowing); ft³/s, cubic feet per second; mg/L, milligrams per liter; µS/cm, microsiemens per centimeter; °C, degrees Celsius; µg/L, micrograms per liter; lb/d, pounds per day; E, estimated, <, less than; --, no data]

Site number	Discharge, instantaneous (ft ³ /s) ¹		Oxygen, dissolved (mg/L)		Field pH (standard units)		Specific conductance (µS/cm)		Temperature (°C)		Net alkalinity (mg/L CaCO ₃)		Chloride, dissolved (mg/L)		Sulfate, dissolved (mg/L)		Aluminum, dissolved (µg/L)		Iron, dissolved (µg/L)		Manganese, dissolved (µg/L)		Load, aluminum, iron, and manganese dissolved (lb/d)	
	Jun	Oct	Jun	Oct	Jun	Oct	Jun	Oct	Jun	Oct	Jun	Oct	Jun	Oct	Jun	Oct	Jun	Oct	Jun	Oct	Jun	Oct	Jun	Oct
1	0.4	0.85	6.0	9.0	6.4	5.8	386	303	22.5	11.0	2.10	6.80	99.0	71.0	22.0	24.0	13.0	88.0	21.0	28.0	230	340	0.58	2.1
2	E.10	.13	9.4	9.9	6.9	6.6	293	283	15.0	9.50	11.0	14.0	71.0	67.0	9.90	13.0	E7.00	<15.0	15.0	15.0	24.0	18.0	.02	.03
3	.49	1.3	9.3	11	7.1	6.4	316	298	18.0	8.30	9.50	7.60	83.0	69.0	16.0	21.0	11.0	20.0	17.0	17.0	17.0	110	.11	1.0
10	.10	<01	7.7	9.5	6.4	6.3	50	44	14.5	9.80	9.50	9.80	1.30	1.30	9.50	11.0	30.0	26.0	65.0	33.0	18.0	10.0	.07	<01
13	.03	.39	4.7	8.7	6.6	6.6	42	30	23.0	14.5	12.0	11.0	.840	.930	7.10	6.40	E7.00	E8.00	21.0	200	63.0	110	.01	.67
18	--	<01	--	9.1	--	3.5	--	366	--	9.80	--	-35.0	--	15.0	--	96.0	--	2,900	--	1,300	--	570	--	.26
22	1.4	3.1	6.4	--	6.1	6.3	1,470	1,460	14.0	14.0	6.70	-19.0	16.0	20.0	740	710	23.0	E15.0	37,000	43,000	4,900	5,200	320	800
23	1.8	8.0 ³	9.4	--	6.8	6.4	1,380	1,430	20.0	14.0	40.0	3.70	19.0	20.0	690	720	17.0	<15.0	3,500	26,000	4,200	5,300	73	1,350
28	2.0	2.9	9.6	5.6	6.9	7.0	1,160	1,070	20.0	13.0	245	211	35.0	36.0	320	280	<10.0	<15.0	4,400	7,000	4,000	3,700	90	170
29	² E3.8	8.2	--	8.9	--	6.8	--	1,390	--	15.0	--	18.0	--	22.0	--	690	--	<15.0	--	15,000	--	4,900	180	880
35	--	E.07	<05	2.2	6.7	6.9	1,530	1,440	13.0	13.0	520	655	5.00	4.80	350	320	E9.00	<15.0	9,200	1,100	1,900	1,600	--	1
37	6.9	--	5.4	--	4.6	--	1,430	--	21.5	--	-36.0	--	8.50	--	800	--	E9.00	--	12,000	--	7,100	--	710	--
38	6.9	4.0	3.0	1.1	5.0	4.3	1,450	1,390	19.5	14.2	-42.0	-38.0	8.50	8.10	840	800	<10.0	16.0	16,000	13,000	7,000	6,800	860	430
39	3.9	.54	7.5	10	3.5	3.4	1,530	1,440	23.0	13.0	-32.0	-48.0	8.00	7.70	860	820	320	2,500	1,100	710	6,700	7,100	170	30
41	--	E.02	--	3.0	--	4.0	--	615	--	14.5	--	-47.0	--	57.0	--	210	--	6,700	--	390	--	2,500	--	1
42	<01	.81	3.3	10	4.2	5.6	205	86	18.0	11.0	--	4.50	--	4.80	--	28.0	--	180	--	E8.00	--	190	--	1.7
43	.10	.89	8.4	11	3.0	3.3	761	378	18.0	11.0	-99.0	-68.0	18.0	5.30	240	160	6,100	5,600	5,700	4,900	2,700	1,700	7.9	59
44	.02	E.31	8.0	--	6.4	6.7	163	110	19.0	12.5	25.0	13.0	26.0	17.0	8.00	11.0	<10.0	<15.0	56.0	200	58.0	78.0	.02	.49
45	.20	1.2	7.9	9.0	3.5	3.5	733	642	16.0	12.5	-51.0	-69.0	30.0	11.0	320	350	3,100	6,900	6,500	4,600	3,600	3,800	14	100
46	.20	E.01	7.8	8.6	6.6	3.6	154	722	18.0	11.0	16.0	--	19.0	--	18.0	--	33.0	--	55.0	--	23.0	--	0.12	--
47	--	1.2	.50	9.6	6.4	3.6	636	737	21.0	9.40	--	-58.0	--	14.0	--	340	--	6,000	--	2,900	--	3,500	--	81
48	.49	<1.2	.30	11	6.4	3.6	540	663	20.0	9.50	45.0	-52.0	35.0	14.0	150	300	27.0	5,300	9,200	2,200	1,500	3,100	28	68
49	4.1	1.8	7.2	10	3.6	3.6	1,480	887	23.5	10.5	-29.0	-52.0	9.60	12.0	840	440	350	5,000	1,400	2,000	6,800	4,600	190	110
50	7.2	³ 8.3	<05	.30	6.0	6.1	1,720	1,720	15.0	15.2	42.0	12.0	16.0	14.0	840	890	14.0	<15.0	55,000	55,000	5,200	5,300	2,300	1,540
51	16	9.7	7.0	8.8	6.3	6.5	1,560	1,450	17.5	14.0	-63.0	-41.0	13.0	14.0	110	770	29.0	71.0	29,000	37,000	6,100	5,700	3,030	2,200

¹Represents instantaneous flow rate unless otherwise indicated.

²Estimated from upstream sites.

³Daily mean flow rate substituted for instantaneous flow rate.

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The presence of coliform bacteria in the stream-water samples is consistent with other observations implying that raw sewage from combined sewer overflows, leaky sewer pipes, residences, or other sources could have been a source of contamination. Likewise, the presence of ammonia is characteristic of raw-sewage contamination because ammonia is eliminated when oxidized to nitrite and then nitrate during treatment. Elevated concentrations of nitrate may indicate additional contamination from treated sewage or some other source such as nitrogen-containing fertilizers. Sewage commonly is cited as a source of pathogens like bacteria, viruses, protozoans, fungi, and parasites (Town, 2001), but it also introduces alkalinity that can moderate pH (fig. 17) and may influence the concentration or speciation of dissolved metals.

Sewage contamination and the associated alkalinity it introduces may have been the cause of pronounced changes in

pH near the mouth of South Branch (site 48). The pH distribution at site 48 is distinctly bimodal (fig. 17). On June 24, 1999, a time when the concentration of ammonia was relatively high (1.3 mg/L) and *E. coli* bacteria were present (120 col/100 mL), the pH was 6.6 and excess alkalinity was available to buffer protons (net alkalinity range was 41.0 mg/L CaCO₃). On October 7, 1999, sewage contamination was not great enough to appreciably alter water chemistry; the concentration of ammonia was only 0.17 mg/L and the water was acidic (pH was 3.6; net alkalinity was -52.0 mg/L CaCO₃) (fig. 17). The presence of *E. coli* on June 24, 1999, suggests that mixing sewage with acidic AMD can moderate pH, but bacteria and associated pathogens can persist. Further study is needed to determine bacteria and nitrogen sources and to evaluate the fate of sewage-related pathogens upon mixing with AMD.

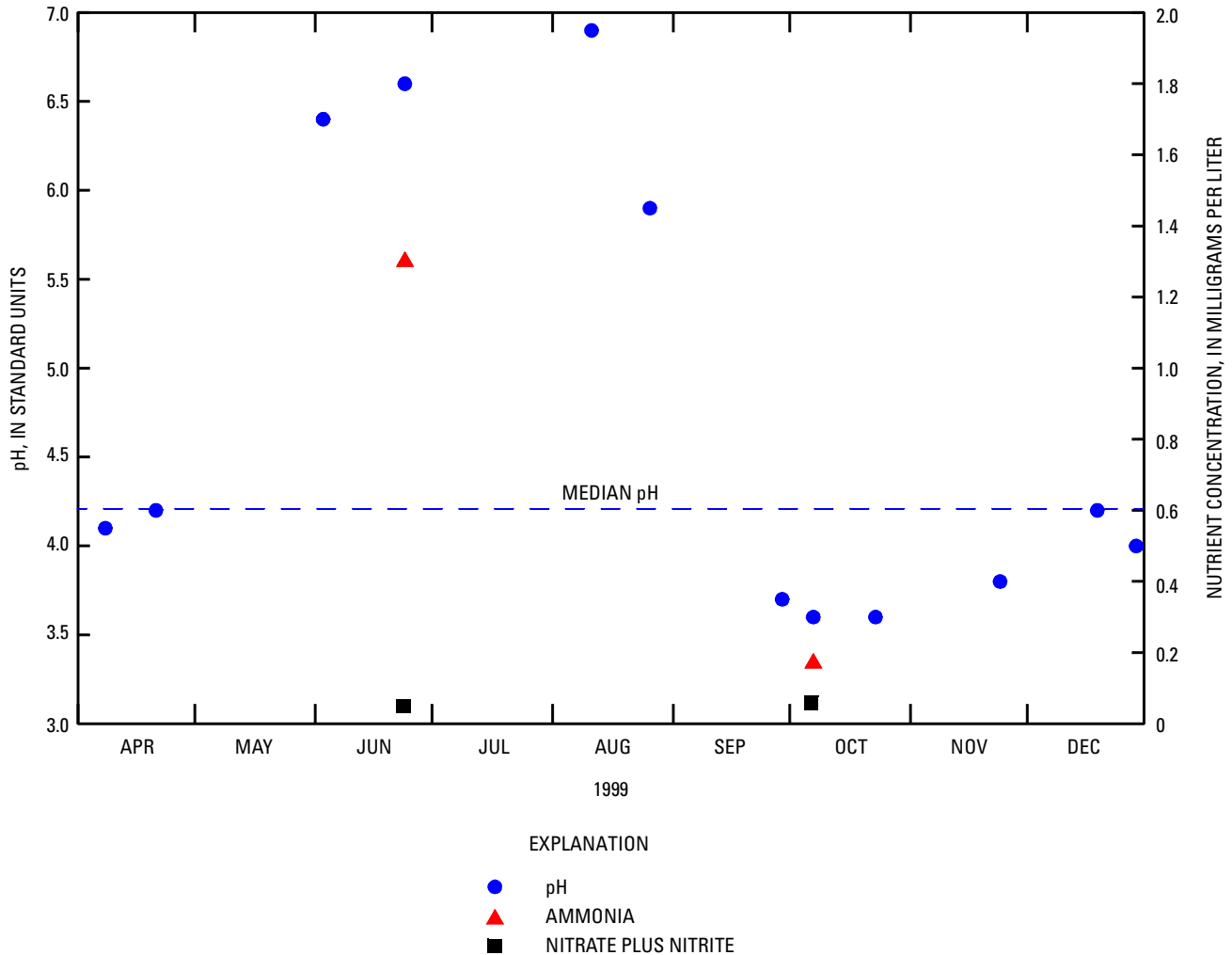


Figure 17. Variation of pH and selected nitrogen species in stream water approximately 300 feet upstream from the mouth of South Branch (site 48), Luzerne County, Pennsylvania.

Variations in the quality of water in Nanticoke Creek, especially in the reach downstream of the Truesdale Mine Discharge, indicate probable effects from sewage contamination, but large AMD inputs from the Truesdale Mine have a greater influence on the chemistry of the downstream reach. During normal base-flow conditions (flow rates of about 1 to 10 ft³/s) at site 23, the majority of flow originated from the Truesdale Mine Discharge as poorly oxygenated, iron-laden AMD with pH greater than 6. However, as below-normal precipitation (fig. 10) and high temperatures lead to drought conditions during the summer of 1999 (figs. 11 and 12), the Truesdale Mine

stopped discharging, and streamflow in Nanticoke Creek decreased to less than 0.1 ft³/s. Along with the decrease in flow, pH declined from near-neutral to less than 4.0 and SC increased downstream at site 23 (fig. 18). Increased SC associated with these low pH values can be attributed to the release of protons combined with evaporation and the possible release of additional sulfate and acidity from renewed oxidation of previously inundated pyrite. The discussion below describes how iron oxidation within the stream channel may account for these observed trends in water quality.

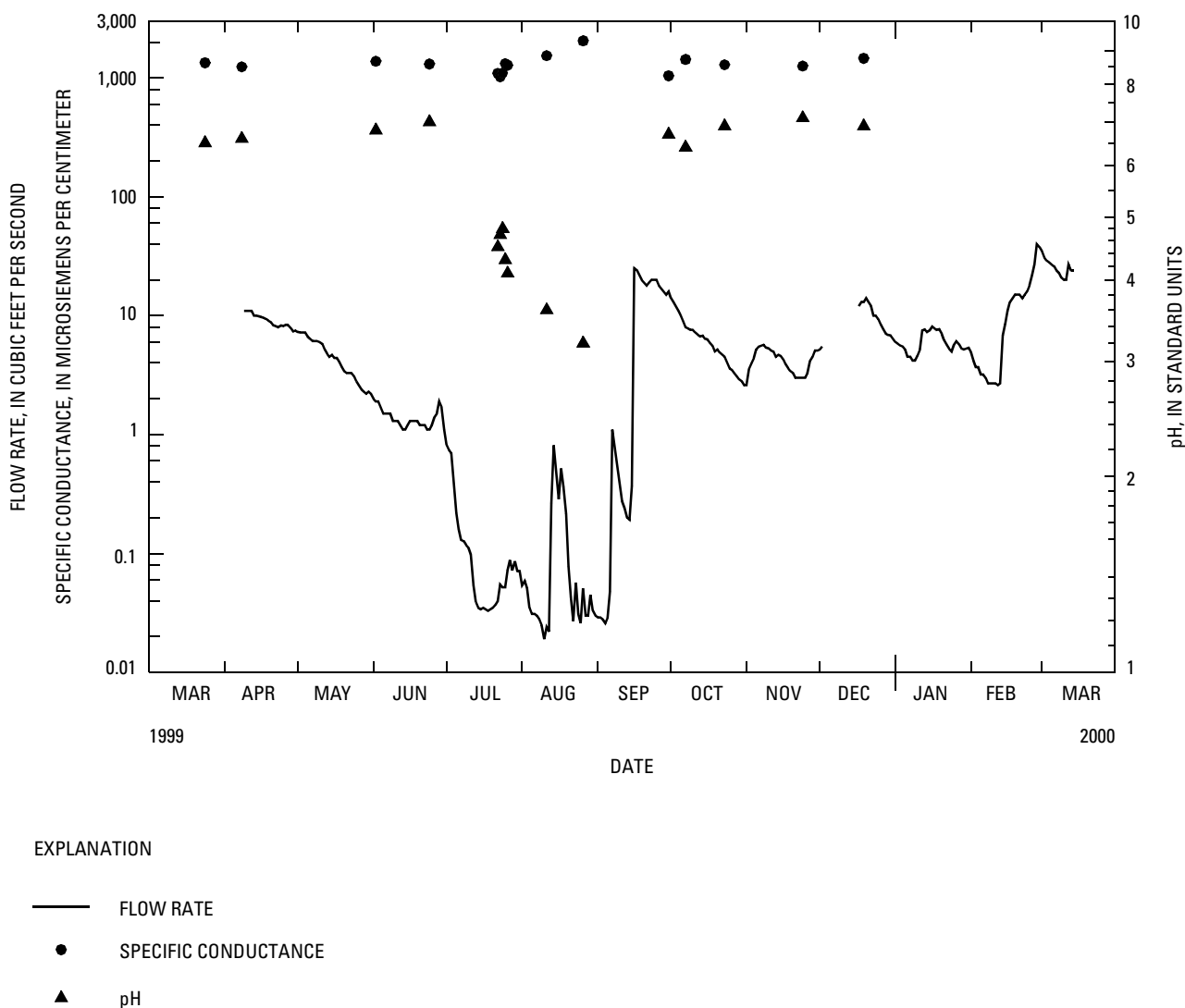
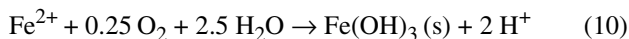


Figure 18. Variability of pH and specific conductance over a range of flow conditions in Nanticoke Creek downstream of Truesdale Mine Discharge (Dundee Outfall) (site 23), Luzerne County, Pennsylvania.

Effects of Iron Oxidation and Hydrolysis on Water Quality

The large AMD sources in the Nanticoke and Newport Creek Basins were poorly oxygenated and had elevated concentrations of dissolved iron. Despite initial pH values that can be near-neutral (pH of 6.0 to 8.0), samples containing dissolved iron ultimately can have acidic pH (< 4.5), as indicated by negative values for net alkalinity (table 5), because of the combined effects of the oxidation of Fe^{2+} to Fe^{3+} (eq. 3) and the consequent precipitation of $\text{Fe}(\text{OH})_3$ (s) (eq. 4):



As indicated by the overall reaction shown in equation 10, the oxidation and hydrolysis of dissolved iron releases H^+ . Hence, the pH will decline unless the sample pH is buffered by excess alkalinity. As explained in detail by Kirby and Cravotta (2004), the pH for net-alkaline samples will remain near-neutral, but that for net-acidic samples ultimately will decline to acidic values as equation 10 proceeds to completion. Although equation 10 is thermodynamically feasible, the rate of Fe^{2+} oxidation under field conditions in mine-drainage settings is complex and can vary as a function of temperature, pH, DO, Fe^{2+} concentration, and bacterial activity (Singer and Stumm, 1970; Stumm and Morgan, 1996; Kirby and Elder-Brady, 1998; Kirby and others, 1999). The rate of Fe^{2+} oxidation commonly is rate-limiting; Fe^{3+} hydrolysis and precipitation are nearly instantaneous (Singer and Stumm, 1970; Stumm and Morgan, 1996).

During normal base-flow conditions, with flow rates of 1 to 10 ft^3/s , estimated traveltimes from the Truesdale mine discharge (site 22) to site 23 on Nanticoke Creek were 2.9 to 0.7 hours, respectively (fig. 19A). In contrast, during extreme low-flow conditions, with flows less than 0.1 ft^3/s , traveltimes were greater than 15 hours from site 22 to site 23 (fig. 19A). During these low-flow conditions, the pH at site 23 decreased to acidic values as low as 3.2 with corresponding concentrations of dissolved iron ranging from less than 1 to 5 mg/L (figs. 19C and 19D). The acidic pH values can be attributed, all or in part, to the prolonged traveltimes and, accordingly, more extensive oxidation and hydrolysis of initially dissolved iron.

When flowing, the quality of water discharged from the Truesdale Mine Discharge (site 22) ranged from net alkaline to net acidic but with relatively constant pH of 5.9 to 6.3 and concentrations of iron of 36 to 43 mg/L during the study (table 5; appendix 2). After discharging to the surface, DO increased rapidly because of exchange with the atmosphere. The DO in Nanticoke Creek downstream of the Truesdale Mine Discharge (site 23) ranged from 6.0 to 9.4 mg/L (table 5; appendix 2). During normal and high base-flow conditions, with discharges greater than 1 ft^3/s , the quality of water in Nanticoke Creek at site 23 exhibited increased pH of 6.4 to 7.0 but decreased concentrations of iron of less than 1 mg/L to 30.3 mg/L compared to the Truesdale Mine Discharge at site 22 (fig. 19B). The decreased concentrations of dissolved iron at site 23 compared to site 22 could result from varying degrees of iron oxidation

and hydrolysis within the stream channel—essentially a function of traveltime (reaction time) from site 22 to site 23. Increased pH at site 23 during high base-flow conditions can be attributed to the rapid loss of dissolved carbon dioxide compared to a much slower rate of Fe^{2+} oxidation.

Various kinetic oxidation models that considered effects of pH, DO, and Fe^{2+} concentration were used to evaluate the effects of iron oxidation and hydrolysis on the quality of water within the Nanticoke Creek channel (figs. 19B, 19C, and 19D). Stumm and Lee (1961) proposed a rate law that is commonly cited for the abiotic oxidation of Fe^{2+} under laboratory conditions:

$$d[\text{Fe}^{2+}]/dt = -k [\text{Fe}^{2+}][\text{OH}^-]^2 \text{Po}_2 \quad (11)$$

where $[\text{Fe}^{2+}]$ and $[\text{OH}^-]$ are concentration in moles per liter (mol/L), Po_2 is partial pressure of atmospheric oxygen (O_2) (atm), and k is the rate constant, 1.33×10^{12} square liters per square mole per atmosphere per second [$\text{L}^2/(\text{mol}^2 \times \text{atm} \times \text{s})$]. The rate model indicated by equation 11 was incorporated in the geochemical program PHREEQC (Parkhurst and Appelo, 1999, example 9).

Williamson and others (1992) reported that the biological oxidation rate for Fe^{2+} in surface waters receiving AMD is both faster than the abiotic oxidation rate indicated by equations 11 and 12 and independent of pH over the range 2.5 to 6.0, as indicated by the differential rate expression:

$$d[\text{Fe}^{2+}]/dt = -k' [\text{Fe}^{2+}], \quad (12)$$

where the rate constant is $1 \times 10^{-3.61}/\text{s}$. Cravotta and Trahan (1999) used the above model to describe the removal of dissolved iron from an OLD.

For this study, the above oxidation models were evaluated by solving the integrated form of equations 11 or 12:

$$[\text{Fe}]_t = [\text{Fe}]_{t=0} \exp\{-k' \times t\} \quad (13)$$

where k' includes specified constant values of pH and O_2 . For specified pH and elapsed time, t , the models computed the concentration of residual Fe^{2+} for an initial concentration of Fe^{2+} of 40 mg/L and constant DO of 7.37 mg/L or Po_2 of 0.21 atm. The models did not consider the effects of iron hydrolysis on the concentration of dissolved iron or pH. Hence, the PHREEQC model (example 9) was modified to consider various oxidation-rate models and solubility control by $\text{Fe}(\text{OH})_3$ (s) with water-quality data for the Truesdale Mine (site 22). The results of simulations for each of the various models are summarized in figures 19C and 19D.

The oxidation models indicate a wide range of results (figure 19B), few of which are in agreement with measured data (figs. 19C and 19D). Although the oxidation-rate models that assume constant pH of 6.5 (models) yield results for residual iron that are comparable to measured values, they are not realistic models for the measured pH, which varied by nearly 4 units. The PHREEQC models that considered pH as a variable

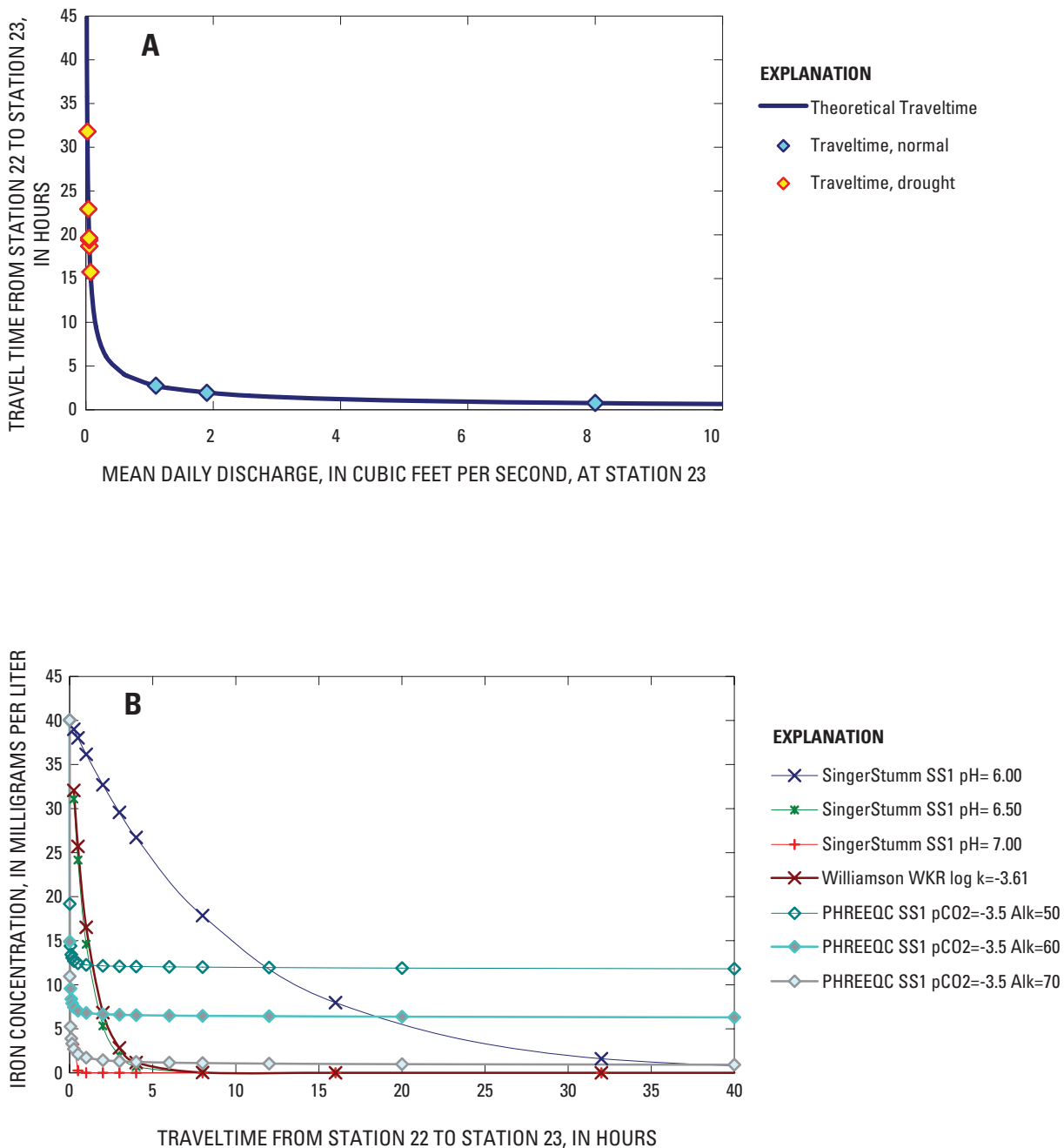


Figure 19. Relations between streamflow in Nanticoke Creek downstream of the Truesdale Mine Discharge (Dundee Outfall) (site 23), traveltime from the Truesdale Mine Discharge (site 22) approximately 2,370 feet upstream to site 23, and the pH and iron concentration at sites 22 and 23 during 1999.

[A, Estimated traveltime from site 22 to site 23 and associated measured values of flow rate, pH, and iron concentration for normal to high base-flow and drought conditions; B, Various models of iron-oxidation kinetics, assuming initial iron concentration of 40 mg/L, oxygen saturation, and a range of pH and alkalinity values (“SingerStumm (SS1)” after Singer and Stumm (1970); “Williamson WKR” after Williamson and others (1992); PHREEQC SS1 uses rate model of Singer and Stumm (1970) and PHREEQC WKR uses rate model of Williamson and others (1992) after example 9 of Parkhurst and Appelo (1999); C and D, Measured and simulated concentrations of dissolved Fe²⁺ and pH as a function of traveltime, Fe²⁺ oxidation rate, and initial alkalinity assuming equilibrium with atmospheric O₂ and CO₂ and solubility control by amorphous Fe(OH)₃ (s)].

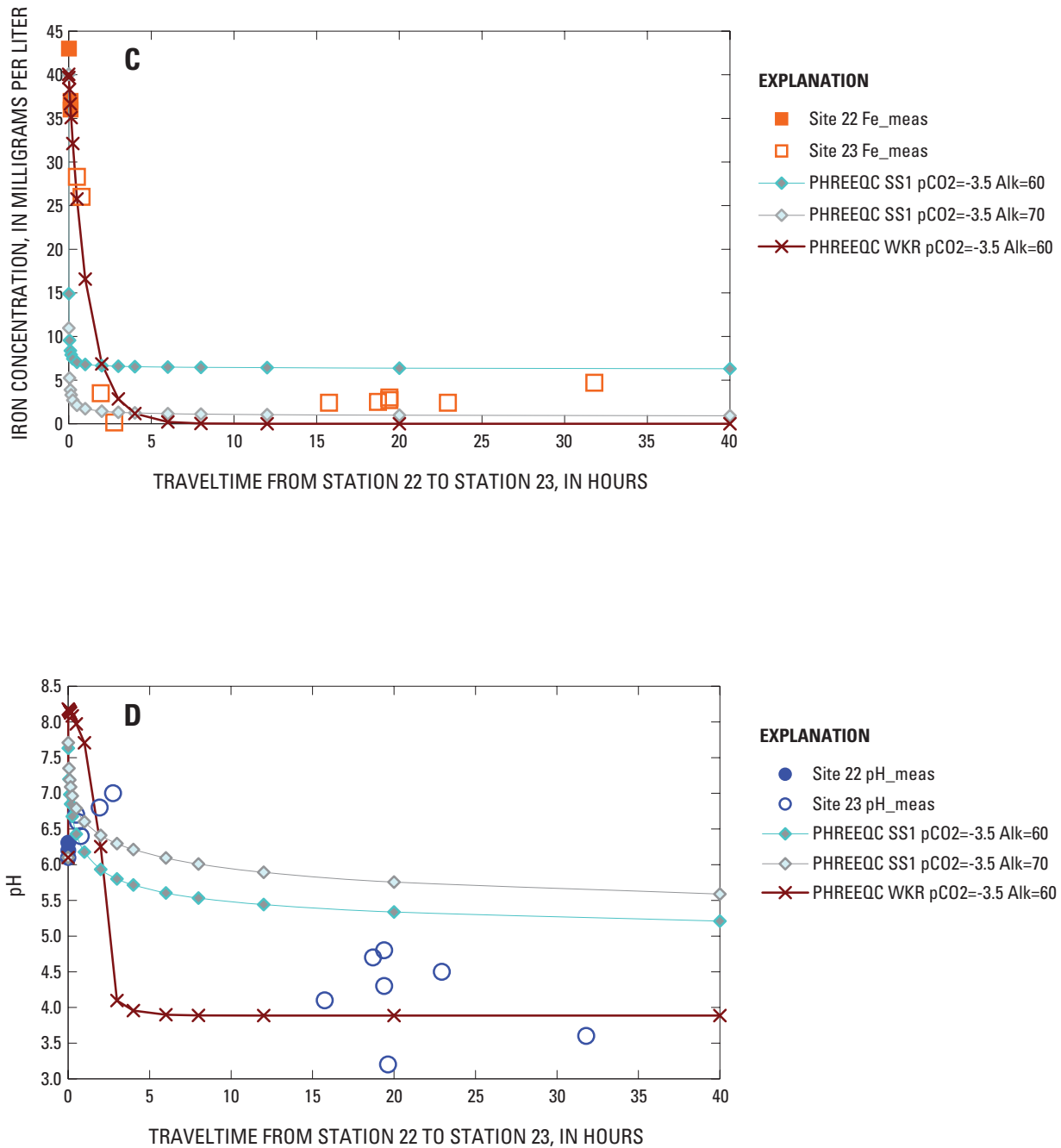


Figure 19. Relations between streamflow in Nanticoke Creek downstream of the Truesdale Mine Discharge (Dundee Outfall) (site 23), traveltime from the Truesdale Mine Discharge (site 22) approximately 2,370 feet upstream to site 23, and the pH and iron concentration at sites 22 and 23 during 1999.—Continued

[A, Estimated traveltime from site 22 to site 23 and associated measured values of flow rate, pH, and iron concentration for normal to high base-flow and drought conditions; B, Various models of iron-oxidation kinetics, assuming initial iron concentration of 40 mg/L, oxygen saturation, and a range of pH and alkalinity values (“SingerStumm (SS1)” after Singer and Stumm (1970); “Williamson WKR” after Williamson and others (1992); PHREEQC SS1 uses rate model of Singer and Stumm (1970) and PHREEQC WKR uses rate model of Williamson and others (1992) after example 9 of Parkhurst and Appelo (1999); C and D, Measured and simulated concentrations of dissolved Fe²⁺ and pH as a function of traveltime, Fe²⁺ oxidation rate, and initial alkalinity assuming equilibrium with atmospheric O₂ and CO₂ and solubility control by amorphous Fe(OH)₃ (s)].

in the Fe^{2+} oxidation kinetics, solubility control of Fe^{3+} by $\text{Fe}(\text{OH})_3$ (s), and solution equilibrium with atmospheric O_2 and CO_2 yielded realistic results (figs. 19C and 19D). The abiotic model of Singer and Stumm (1970), considering initial alkalinity of 60 or 70 mg/L, yielded concentrations of dissolved iron that were comparable to measured values but with pH values larger than measured values. The actual rate of Fe^{2+} oxidation appeared to be faster than the modeled rate. For the same conditions, only considering the faster “biological” oxidation rate model of Williamson and others (1992), predicted results for Fe^{2+} and pH generally were comparable to the observed trends. Thus, as described in more detail by Kirby and others (1999), a combination of abiotic and biological factors can contribute to the oxidation of Fe^{2+} under field conditions.

The relation between the traveltime and the corresponding measured pH and rates of iron removal between sites 22 and 23 on Nanticoke Creek and the associated oxidation kinetic models described above have implications for the design or optimization of AMD treatment systems. Generally, iron removal was faster for conditions where alkalinity was sufficient to buffer the pH to be near-neutral. These conditions were present during normal base-flow conditions 1.0 to 10 ft^3/s , with traveltimes less than 3 hours (figs. 19A, 19C, and 19D). However, with the decline in pH to acidic values during low base-flow conditions (< 1.0 ft^3/s), efficiency of iron removal declined. This latter condition, particularly during drought, could be mitigated with as little as 10 mg/L additional alkalinity to buffer the pH to be near-neutral. In a study of passive mine-drainage treatment systems, Kirby and others (1999) found that from pH 5 to 6.4, iron oxidation was positively correlated with pH and that treatment efficiency could be improved by increasing the pH within this range, at which oxidation is primarily abiotic. Aerobic wetlands or ponds to treat AMD from the Truesdale Mine or similar AMD could include components such as limestone in the substrate to produce the alkalinity needed to buffer the pH and sustain high rates of abiotic Fe^{2+} oxidation. Alternatively, limestone or other alkaline-producing materials could be applied to streambeds or spoil banks.

Concentrations of Metals

Stream water above the mined areas (sites 1, 2, 3, 10, 13, and 14) had low concentrations of dissolved iron (less than or equal to 200 $\mu\text{g}/\text{L}$; appendix 2) but AMD discharging from lower points in the valley often greatly exceeded the 7,000 $\mu\text{g}/\text{L}$ limit for AMD established in SMCRA (Commonwealth of Pennsylvania, 1998a, 1998b). AMD discharging from the Truesdale Mine at site 22 had a median concentration of dissolved iron of 37,000 $\mu\text{g}/\text{L}$ compared to a median concentration of 56,400 $\mu\text{g}/\text{L}$ for the Susquehanna Number 7 discharge at site 50 (appendix 2). AMD at the mine-pit overflow near Newport Creek at site 38 had concentrations of 16,000 $\mu\text{g}/\text{L}$ in June 1999 and 13,000 $\mu\text{g}/\text{L}$ in October 1999 (table 5). As a result of these AMD inputs, receiving streams in both basins had concentrations of dissolved iron that were greater than those estab-

lished to meet the CWF designation (300 $\mu\text{g}/\text{L}$; Commonwealth of Pennsylvania, 2002).

Concentrations of dissolved iron in Nanticoke Creek were appreciably greater when the Truesdale Mine Discharge was flowing. Stream water in Nanticoke Creek, approximately 0.5 mi downstream of the Truesdale Mine Discharge (site 23), had a median concentration of dissolved iron of 4,700 $\mu\text{g}/\text{L}$ (see appendix 2). During the period of below-normal precipitation between June and early September 1999 (fig. 10), the Truesdale Mine Discharge was only flowing periodically and concentrations of dissolved iron at site 23 were below 3,500 $\mu\text{g}/\text{L}$, except for one value of 4,700 $\mu\text{g}/\text{L}$ on August 11, 2000. After rainfall from remnants of Hurricane Floyd recharged the mine pool in mid-September 1999 (figs. 11 and 12), concentrations of dissolved iron exceeded 5,000 $\mu\text{g}/\text{L}$ and were greater than or equal to 25,000 $\mu\text{g}/\text{L}$ in five of seven samples (appendix 2).

In the Newport Creek Basin, concentrations of dissolved iron were greatest in AMD from the Susquehanna Number 7 Mine (55,000 $\mu\text{g}/\text{L}$ in June and October 1999) and least in a small discharge at site 45 near Sheatown, Pa. (6,500 $\mu\text{g}/\text{L}$ in June and 4,600 $\mu\text{g}/\text{L}$ in October; table 5). Intermediate concentrations were measured at the mine-pit overflow in the headwaters of Newport Creek (16,000 $\mu\text{g}/\text{L}$ in June and 13,000 $\mu\text{g}/\text{L}$ in October). Attenuation of iron supplied by the mine-pit overflow to Newport Creek resulted in lower concentrations at site 39, which is about 1.25 mi downstream; concentrations of dissolved iron were 9,200 $\mu\text{g}/\text{L}$ in June and 2,200 $\mu\text{g}/\text{L}$ in October.

As described for Nanticoke Creek, attenuation of dissolved iron between sites 38 and 39 on Newport Creek and in some other stream reaches also can be explained from oxidation of Fe^{2+} to Fe^{3+} and the subsequent formation of $\text{Fe}(\text{OH})_3$ (s). The reach between sites 38 and 39 loses stream water as it flows through a wide flat area with relatively slow velocities and a long residence time. Wetlands with similar attributes such as those constructed to treat a portion of the Truesdale Mine Discharge (Klemow, 2002b, 2003) promote oxidation of dissolved iron and precipitation of $\text{Fe}(\text{OH})_3$ (s) that can be settled from the water column at rates averaging 20 (g/m^2)/d (180 (lb/acre)/d) (Hedin and others, 1994). Even though base-flow samples collected for this study showed a decrease in concentration of dissolved iron between sites 38 and 39, concentrations during stormflows were poorly characterized. $\text{Fe}(\text{OH})_3$ (s) on the streambed may be resuspended and distributed downstream during stormflows such as those experienced during remnants of Hurricane Floyd (fig. 12).

Concentrations of dissolved aluminum were elevated in AMD entering Newport and Nanticoke Creek, but in-stream concentrations varied locally, exceeding the limit for CWF use at some locations and times. A small AMD discharge (less than 0.10 ft^3/s) below the confluence of Leuder Creek and Nanticoke Creek at site 20 had the highest concentrations of dissolved aluminum; 10,900 $\mu\text{g}/\text{L}$ on June 15, 1999, and 38,600 $\mu\text{g}/\text{L}$ on September 20, 2000 (appendix 2). This site was not part of the basin-wide synoptic sampling in June and October 1999. In Newport Basin, a small AMD discharge at site 45 near the headwaters of South Branch had 6,900 $\mu\text{g}/\text{L}$ of dissolved aluminum

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during synoptic sampling in October 1999 (table 5). This source and other sources in the upstream reach of South Branch containing sites 41 (dissolved aluminum concentration was 6,700 $\mu\text{g/L}$ in October 1999) and 43 (5,600 $\mu\text{g/L}$ in October 1999) contributed metals to South Branch and Newport Creek when flowing. Variation in concentrations of dissolved aluminum near the mouth of South Branch seemed to be related to ephemerally flowing sewage and AMD sources, and the pH of stream water.

When AMD at sites 41, 45, or other nearby sources was flowing and pH in South Branch remained acidic (less than 4.5), concentrations of dissolved aluminum remained elevated at the mouth (site 48). For example, in October 1999, stream water at site 43 (near Alden, Pa.) and at site 48 (near the mouth of South Branch) had pH less than 4.0 and concentrations of dissolved aluminum greater than 5,000 $\mu\text{g/L}$ (table 5). However, in June 1999, stream water at site 48 was near-neutral (pH was 6.4) and the concentration of dissolved aluminum was only 27 $\mu\text{g/L}$ despite corresponding lower pH (3.0) and higher concentrations of dissolved aluminum (6,100 $\mu\text{g/L}$) upstream at site 43. The elevated pH at site 48 was probably associated with alkalinity from sewage inflows between sites 43 and 48 that caused a decrease in dissolved aluminum through dilution and precipitation of aluminum hydroxide ($\text{Al}(\text{OH})_3$ (s)). The “gray” appearance of the stream during the June 1999 sampling event can be attributed to transport of suspended $\text{Al}(\text{OH})_3$ (s) along with particulate matter from the sewage. Infiltration through the streambed upstream of site 48 and dilution by uncontaminated water from Fairchild Pond and a suspected water-supply leak at site 46 near Sheatown, Pa., may also have played a role in attenuation of dissolved aluminum in South Branch.

Other undocumented sources of aluminum to Newport Creek exist between the acidic mine-pit overflow at site 38 and site 39, which is just upstream of South Branch. During synoptic sampling in October 1999, the concentration of dissolved aluminum of the mine-pit overflow was only 16 $\mu\text{g/L}$, but concentrations of aluminum increased downstream at site 39 to 2,500 $\mu\text{g/L}$. The source(s) of aluminum was not sampled but probably is a diffuse seep(s) into one of the wetlands downstream of site 38. Regardless, the dissolved aluminum in water at site 39 was not transported conservatively downstream. When Newport Creek mixed with net alkaline AMD from site 50 (Susquehanna Number 7 Mine discharge), most dissolved aluminum was converted to $\text{Al}(\text{OH})_3$ (s) and may have settled from the water column. This conversion coupled with dilution caused concentrations of dissolved aluminum to be low at the mouth of Newport Creek, less than 100 $\mu\text{g/L}$ during sampling events in June and October 1999.

Manganese concentrations were less than 500 $\mu\text{g/L}$ in headwater locations that did not receive AMD from the mine workings (table 5; appendix 2) but were elevated downstream. The largest concentrations of manganese were in AMD from the Truesdale Mine (site 22) in Nanticoke Basin (range was 3,600 to 5,200 $\mu\text{g/L}$), site 38 at the mine-pit overflow in Newport Basin (range was 6,800 to 7,000 $\mu\text{g/L}$), sites 41 (2,500 $\mu\text{g/L}$ in October 1999) and 45 (range was 3,600 to 3,800 $\mu\text{g/L}$) on

South Branch, and site 50 (range was 5,100 to 5,300 $\mu\text{g/L}$) at the Susquehanna Number 7 Mine discharge near the mouth of Newport Creek. As expected for acidic to near-neutral pH conditions, dissolved manganese was transported conservatively in the streams and any changes in concentration were the result of dilution or additional manganese from contributing AMD sources.

Presently (2007), small-scale passive-treatment systems are in place to remove iron, aluminum, and to a lesser extent manganese from a small unsampled AMD discharge to Espy Run (Klemow, 2002b) and a portion of the Truesdale Mine Discharge (Klemow, 2003). Passive-treatment technologies such as those developed in the mid-1980s and 1990s to treat AMD can reduce the costs of infrastructure, chemical amendments, and maintenance compared to conventional active treatment (Hedin and others, 1994). Wetlands and other passive-treatment strategies may be appropriate to treat other AMD sources in the study area and the remainder of AMD from the Truesdale Mine; however, selection of the best strategy for AMD amelioration is highly dependent on pH, DO, and net alkalinity of the water targeted for treatment (Hedin and others, 1994).

Potential Passive-Treatment Strategies for Newport and Nanticoke Creeks

The chemical and flow characteristics for the AMD from selected discharges in the study area were considered with respect to selection criteria for various remedial strategies (fig. 7). Typically, to determine the areal extent of an aerobic wetland system, the average iron-loading rate of the AMD source expressed in grams per day (g/d) is divided by an assumed iron-removal rate of 20 (g/m²)/d based on the work of Hedin and others (1994). However, studies at wetlands receiving AMD have indicated that the prescribed iron-removal rate of 20 (g/m²)/d can underestimate the area needed for effective treatment, particularly as pH and/or concentrations of iron decrease (Tarutis and others, 1999; Kirby and others, 1999). Thus, a range in potential wetland sizes for each AMD source was computed by dividing the maximum iron load (product of maximum flow and maximum concentration of iron for synoptic samples in June and October 1999), in grams per day, by 5 (g/m²)/d (45 (lb/acre)/d) and 20 (g/m²)/d (180 (lb/acre)/d) (table 6).

Although areal iron-removal rates are useful as guidance criteria for estimating wetland size to treat net-alkaline AMD, wetland treatment efficiency can vary greatly because of variable rates of iron oxidation as a function of pH, concentration of iron, alkalinity, temperature, and biological processes (Kirby and others, 1999). The PHREEQC iron-oxidation kinetics model of Parkhurst and Appelo (1999, example 9) using the abiotic rate equation of Singer and Stumm (1970) was adapted to estimate variations in the pH and concentrations of dissolved iron in effluent from the large AMD sources (sites 22, 38, and 50) after exposure to air at the surface. The results of simulations for an elapsed time of 40 hours are illustrated in figure 20 with a reference line for the average detention time of effluent in a wetland of the size specified assuming an iron-removal rate of 20 (g/m²)/d (table 6). Potential pH, and iron-concentration values for the treated effluent from the specified size wetland are indicated by the intersections between the relevant simulated values of pH, and iron (curves) with the dashed vertical line in figure 20. Various results were indicated considering two end-member conditions (equilibrium and disequilibrium) with respect to atmospheric O₂ (pO₂ of 10^{-0.68} atm) and CO₂ (pCO₂ of 10^{-3.5} atm). Under disequilibrium conditions, changes in pH and concentrations of iron tend to be substantially less than those for equilibrium conditions.

As discussed previously, the Truesdale Mine (site 22) effluent ranged from net alkaline to net acidic. For the range of iron-removal rates, a wetland with area of 4 to 16 acres was computed (table 6). The condition simulated in figure 20A considers initial pH of 6.1, alkalinity of 68 mg/L, and dissolved iron of 43 mg/L (net alkalinity of -9.0 mg/L) and shows a reference line for the detention time corresponding to the low end of the wetland area range. The simulation enabled the evaluation of the effect of alkalinity and aeration on the pH and iron-removal efficiency. Under atmospheric disequilibrium conditions, Fe²⁺

oxidation is extremely slow where the pH remains depressed by dissolved CO₂; more than half of the iron (26 mg/L) remains after 40 hours. In contrast, under atmospheric equilibrium conditions, oxidation and precipitation of the dissolved iron are indicated to be initially rapid, with only 5 mg/L remaining after 6 hours. Rapid oxidation is indicated initially because the initial pH increases to 7.6 as CO₂ is decreased to atmospheric equilibrium levels. Within 2 hours, continued oxidation and hydrolysis of remaining dissolved iron causes the pH to decline to values below 6.0. However, removal of the remaining 5 mg/L of dissolved iron is very slow and inefficient because of the relatively slow rate of abiotic Fe²⁺ oxidation at pH values less than 6.0 (Kirby and others, 1999). If Fe²⁺ oxidation were allowed to continue to completion (not shown in figure 20A), the simulation indicates pH could decrease to less than 4.0, and the final alkalinity value would be negative (net acidic). Therefore, additional alkalinity would be needed to buffer the pH to remain near-neutral when conditions are net acidic.

The range of iron-oxidation rates indicated in figure 20A underscores the need to adequately characterize AMD chemistry for a range of hydrologic conditions. For example, although unnecessary to treat the net-alkaline quality commonly observed for site 22 (Klemow, 2002a, 2003), supplemental alkalinity would have been needed during the summer and fall of 1999 when the quality at the Truesdale Mine Discharge and in Nanticoke Creek downstream from the discharge was net acidic (appendix 2; tables 5 and 6). In similar situations, an OLD (passive treatment) or limestone diversion well or lime dosing (active treatment) could be useful to add alkalinity to the AMD source. To achieve a result approaching the atmospheric equilibrium simulation, enhanced aeration may be warranted to drive off CO₂.

The Susquehanna Number 7 Mine (site 50) effluent was consistently net alkaline and, hence, would not warrant a supplemental source of alkalinity for its passive treatment (fig. 7, table 6). The condition simulated in figure 20B considers initial pH of 6.0, alkalinity of 120 mg/L, and dissolved iron of 55 mg/L (net alkalinity of 21.5 mg/L) and can be related to treatment solely by aerobic wetlands. Assuming atmospheric disequilibrium, oxidation is extremely slow, and the pH remains depressed by dissolved CO₂. In contrast, assuming atmospheric equilibrium, oxidation and precipitation of the dissolved iron is rapid, and the pH remains near-neutral. If completely oxidized, the simulation indicates pH could increase from 6.1 to 7.8, and the final alkalinity value would be positive (net alkaline).² Thus, without enhanced aeration to ensure atmospheric equilibrium, a wetland with an area of 12 to 49 acres based on presumed areal removal rates for iron (table 6) could be inadequate. On the other hand, enhanced aeration to

²The water at Susquehanna Number 7 Mine (site 50) in October 1999 had pH of 6.1 when sampled in the field, but after 4 years in storage in the laboratory, the pH had increased to 7.8 (Cravotta and Kirby, 2004). Cravotta and Kirby (2004) described the relations between the pH and net alkalinity of fresh and aged samples.

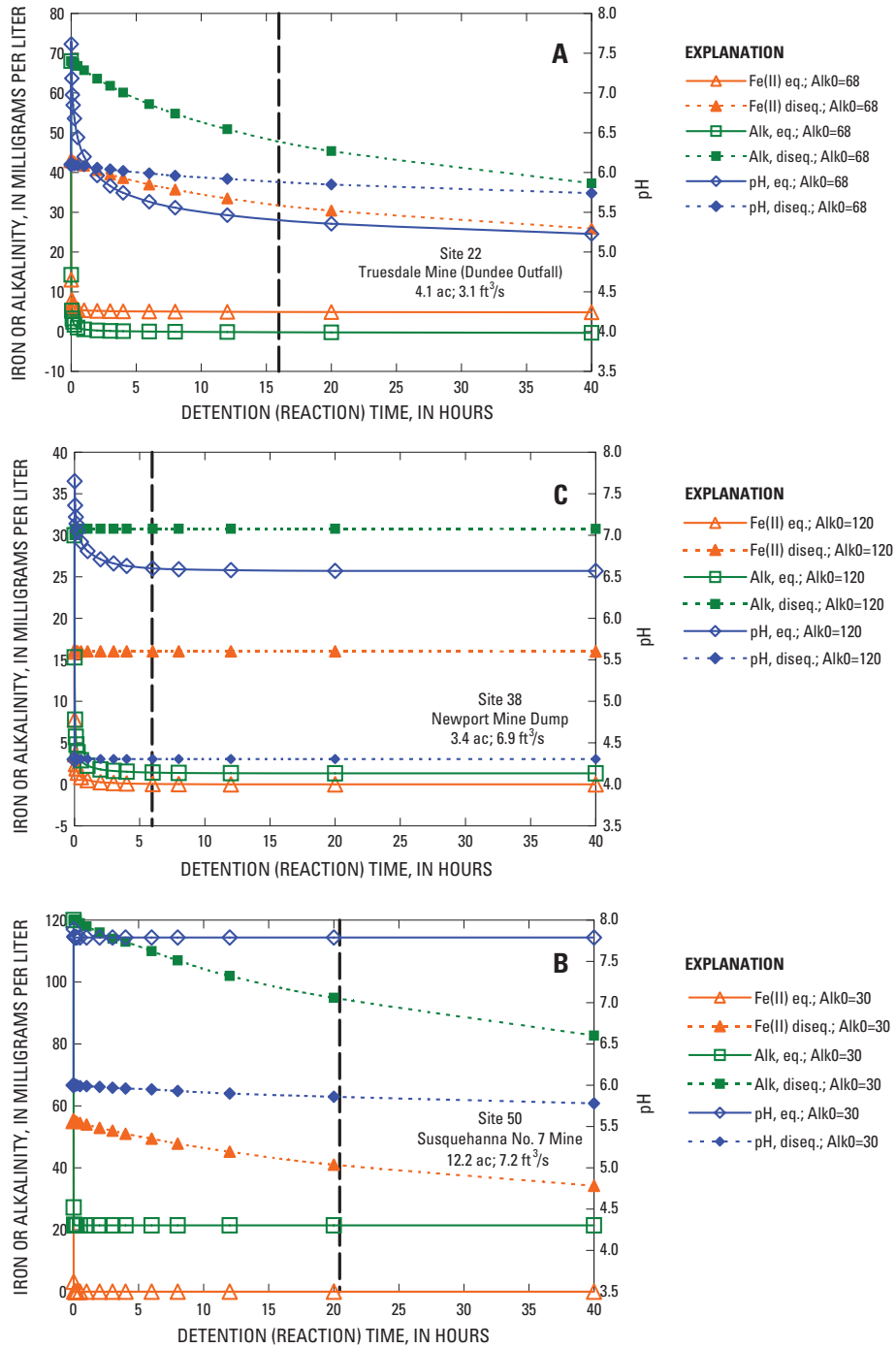


Figure 20. Simulated changes in pH and concentration of dissolved Fe²⁺ in effluent from large AMD sources in Newport and Nanticoke Creek Basins on the basis of PHREEQC iron-oxidation kinetics model (Parkhurst and Appelo, 1999, example 9) using the rate equation of Singer and Stumm (1970) and considering equilibrium with atmospheric O₂ and solubility control by amorphous Fe(OH)₃.

[Open symbols connected by solid curves indicate “equilibrium” conditions, where dissolved CO₂ is in equilibrium with the atmosphere (pCO₂ of 10^{-3.5} atmosphere); solid symbols connected by dashed curves indicate “disequilibrium” conditions, where dissolved CO₂ is not in equilibrium with the atmosphere; ac, acres; ft³/s, cubic feet per second. Vertical dashed line indicates average detention time in a pond or wetland with the area specified, average depth of 1 meter, and maximum flow measured in June or October 1999 (per table 6).]

Table 6. Principal water-quality characteristics in relation to potential remedial strategies¹ for abandoned mine drainage in Newport and Nanticoke Creek Basins, Pennsylvania.

[Remedial strategies are not identified in any order; any treatment design would require additional data and specific analysis; us, upstream; VFCW, vertical-flow compost wetland; ALD, anoxic limestone drain; OLD, flushable oxic limestone drain; OLC, open limestone channel; X, applicable; +, additional; -, not applicable; ?, insufficient data; =, equal to; >, greater than; ≥, greater than or equal to; <, less than; ≤, less than or equal to; ft³/s, cubic feet per second; mg/L, milligrams per liter; (g/m²)/d, grams per meter squared per day; (lb/acre)/d, pounds per acre per day]

Site name and local identification number ²	Principal characteristics ³	Potential remedial strategies ⁴						Comments	Wetland area, acres ⁵	
		Remove culm bank	VFCW	ALD	OLD	OLC	Aerobic pond(s)			Active treatment
Truesdale Mine Discharge (Dundee Outfall) (22)	Intermittent large flow; very high Fe, Mn; low Al; net acidic?; oxic	-	-	-	X	-	+	X	Presently treated with aerobic wetlands; active pumping and enhanced aeration.	4 to 16
Espy Run (28)	Large flow; moderate Fe, Mn; low Al; net alkaline; oxic	-	-	-	-	-	X	-	Presently treated passively with aerobic wetlands.	<1 to 3
Newport Creek Seepage (35)	Small flow; moderate Fe, Mn; low Al; net alkaline; oxic	-	-	-	-	-	X	-	Small size could facilitate passive treatment with aerobic wetlands.	<1
Newport Creek Mine-pit Overflow (38)	Very large flow; very high Fe, Mn; low Al; net acidic; oxic	-	-	-	-	-	-	X	Net-acidic character and large flow volume limit potential for passive treatment.	3 to 14
South Branch Discharge us of Unnamed Tributary (41)	Intermittent small flow; moderate Fe, Mn; very high Al; net acidic; oxic	-	X	-	-	-	+	-	Aluminum may be passively removed with layered compost and limestone system.	<1
Susquehanna Number 7 Mine (50)	Very large flow; very high Fe, Mn; low Al; net alkaline; anoxic	-	-	-	-	-	X	X	Large flow volume; if land available, characteristics suitable for passive treatment with aerobic wetlands.	12 to 49

¹Various remedial strategies and associated water-quality criteria are described in figure 7.

²Site descriptions given in table 1.

³Principal characteristics based on maxima and minima for flow rate and concentrations of alkalinity, dissolved metals, and oxygen (in mg/L) for samples collected in June and October 1999 (table 3). Flow (ft³/s): 'very large' if minimum > 2.0; 'large' if maximum > 1.0 and ≤ 2.0; 'moderate' if maximum > 0.1 and ≤ 1.0; 'small' if maximum ≤ 0.1; 'intermittent' if maximum or minimum = 0. Iron and manganese (mg/L): 'very high' if minimum Fe > 12 and minimum Mn > 4; 'high' if minimum Fe ≤ 12 and minimum Mn ≤ 4; 'moderate' if minimum Fe ≤ 6 and minimum Mn ≤ 2; 'low' if maximum Fe < 3 and maximum Mn < 1. Aluminum (mg/L): 'very high' if maximum > 4; 'high' if maximum > 2; 'moderate' if maximum ≤ 2; 'low' if maximum < 0.2. Net alkalinity (alkalinity - computed acidity; mg/L CaCO₃): 'net acidic?' if maximum ≤ 5; 'net acidic' if maximum ≤ 0; 'net alkaline?' if minimum > 0 or if missing and minimum pH ≥ 6.4; 'net alkaline' if minimum ≥ 5. Dissolved oxygen (mg/L): 'anoxic' if maximum ≤ 1; 'suboxic' if maximum ≤ 2; 'oxic' if minimum > 2.

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

⁵Wetland size range computed by dividing the product of maximum flow rate and maximum iron concentration, in grams per day, by 20 (g/m²)/d (180 (lb/acre)/d) per Hedin and others (1994) and 5 (g/m²)/d (45 (lb/acre)/d). The smaller area is computed using the 20 (g/m²)/d iron-removal rate.

drive off CO₂ could enable effective treatment to remove dissolved iron in wetlands smaller than dictated by the presumed iron-removal rates.

Effluent entering Newport Creek from the mine-pit overflow at site 38 was consistently net acidic, with alkalinity of 0.00 mg/L CaCO₃ and net alkalinity from -42 to -38 mg/L CaCO₃ (table 5). To produce a final pH value near neutrality, an initial concentration of alkalinity of 30 mg/L or greater would be needed. Without this added alkalinity, oxidation of dissolved iron is slow, and as oxidation and hydrolysis proceed, the pH decreases to values less than 4.0.³ For example, the condition simulated in figure 20C considers initial pH of 4.3, alkalinity of 30 mg/L, and dissolved iron of 16 mg/L. As shown by the atmospheric disequilibrium simulation (fig. 20C), addition of 30 mg/L alkalinity does not produce the desired effect on pH and iron concentrations because the dissolved CO₂ maintains low pH values. In contrast, the atmospheric equilibrium simulation (fig. 20C) shows that the combination of 30 mg/L alkalinity and aeration to drive off CO₂ can maintain near-neutral pH and promote rapid oxidation and precipitation of dissolved iron. Given the large flow rate, low pH, and elevated concentrations of DO and iron, an efficient alkalinity-producing system, such as lime dosing (active treatment), followed by a pond or wetland with an area of 3 to 14 acres (table 6) could be considered instead of a passive-treatment system (table 6) to remove the dissolved iron.

Summary

Newport and Nanticoke Creeks drain a combined area of 21.6 mi² (14 and 7.6 mi², respectively) and are mostly within the Northern Anthracite Coal Field in central Luzerne County, Pa. The hydrologic and chemical characteristics of Newport and Nanticoke Basins have been altered by abandoned underground and surface-mining operations that enhance infiltration of surface runoff and stream water and facilitate interbasin transfer of ground water. Headwater streams that flow northward from Penobscot and Little Wilkes-Barre Mountains are unaffected by mining, but once the streams cross mined coalbeds, fractures, and subsidence features in the valley, stream water infiltrates into the mine workings. The water becomes loaded with acidity, metals, and sulfate in the mines. In downstream reaches, the contaminated water resurfaces from fractures or mine openings at topographically low points as abandoned mine drainage (AMD) contaminating the streams but contributing substantially to base flow. Treatment of selected AMD sources using passive methods developed in the mid-1990s could provide a viable option to ameliorate AMD contamination in the study area. Selection of an appropriate treatment strategy would rely on characterization of the range and variability of hydrologic

and chemical properties of the contaminated influent targeted for treatment.

From 1999 to 2000, the U.S. Geological Survey (USGS), in cooperation with Earth Conservancy and in collaboration with Wilkes University, conducted an assessment of Newport and Nanticoke Creek Basins. Assessment field work included (1) continuous stage measurement at 7 sites; (2) synoptic water-quality sampling and flow measurement at 21 sites during June 2-4, 1999, and at 24 sites during October 7-8, 1999; and (3) periodic measurement of flow and water quality at 26 additional sites not included in the synoptic sampling effort. These data were then used to characterize the effects of historical mining and AMD and to identify potential passive-treatment strategies that may ameliorate selected discharges. The data collected for this study represent the type of baseline information needed for documentation of water-quality changes following passive treatment of mine drainage in Pennsylvania and other similar hydrogeologic settings. Because many AMD discharges throughout the Anthracite Coal Region are similar in quality and quantity to those sampled for this investigation, potential remedial strategies presented here may be transferable with little adaptation.

Differences among flows in unmined and mined areas of the Nanticoke and Newport Creek Basins indicate that (1) stream reaches within the mined area and upgradient of large discharges like Truesdale Mine Discharge in Nanticoke Basin and Susquehanna Number 7 Mine discharge near the mouth of Newport Creek generally were either dry or losing reaches; (2) ground water flowing to AMD sites could cross surface-drainage divides; and (3) discharges of AMD in the lower reaches restored flow volumes lost to the mines in the upstream reaches.

The synoptic data, along with additional data collected intermittently, indicate headwaters upstream of the mined area generally were oxygenated (dissolved oxygen concentration range was 4.7 to 11.0 mg/L), near-neutral (pH range was 5.8 to 7.6), and net alkaline (net alkalinity range was 2.0 to 25.0 mg/L CaCO₃), with relatively low concentrations of sulfate (6.40 to 24.0 mg/L) and metals (less than 500 µg/L each of dissolved iron, manganese, and aluminum). The AMD discharges and their downstream receiving waters were characterized by variable flow and water chemistry but generally had elevated concentrations of dissolved metals and sulfate that exceeded limits established by the Surface Mining and Reclamation Act and the Pennsylvania Department of Environmental Protection.

The Truesdale Mine Discharge, located about 0.5 mi upstream of Loomis Park, is the largest AMD source to Nanticoke Creek and was characterized by poorly oxygenated (dissolved oxygen concentration range was 0.80 to 6.40 mg/L) but usually near-neutral flow (pH range was 5.9 to 6.3) that had variable net alkalinity (range was -19.0 to 57.0 mg/L CaCO₃) and that dried up during periods of below-normal precipitation. Concentrations of aluminum were less than 50 µg/L but iron and manganese were elevated (ranges were 30,500 to 43,000 µg/L and 3,600 to 5,200 µg/L, respectively). During low flows (less than 0.1 ft³/s) and when the Truesdale Mine was not

³The water at the Newport Creek mine-pit overflow at site 38 in October 1999 had pH of 4.3 when sampled in the field, but after 4 years in storage in the laboratory, the pH had decreased to 3.4 (Cravotta and Kirby, 2004).

discharging AMD at the borehole, the downstream reach of Nanticoke Creek was acidic (pH between 3.0 and 4.0). These low flows were associated with drought and long travel times (greater than 15 hours), allowing for nearly complete oxidation and hydrolysis of dissolved iron and more extensive production of H^+ ions than under higher flow conditions.

AMD discharges in the upper reaches of the Newport Basin were acidic (pH less than 4.5; net alkalinity less than 0); however, the largest AMD source that entered near the mouth of Newport Creek from the Susquehanna Number 7 Mine was alkaline (pH greater than 5.9; net alkalinity greater than 12.0 mg/L $CaCO_3$). The six AMD discharges sampled in the Newport Basin were poorly oxygenated (dissolved oxygen concentrations less than 3.0 mg/L) except for a discharge near Sheatown, Pa., which had dissolved oxygen concentrations of 7.9 mg/L in June 1999 and 9.0 mg/L in October 1999. The Susquehanna Number 7 Mine discharge had the greatest concentrations of dissolved iron (52,500 to 77,400 $\mu g/L$) but substantial metals contributions also were made by a relatively large mine-pit overflow in the upper reaches of Newport Creek (flows from 4.0 to 6.9 ft^3/s ; dissolved iron concentration from 13,000 to 16,000 $\mu g/L$) and the smaller Sheatown discharge (flows from 0.20 to 1.2 ft^3/s ; dissolved iron concentration from 4,600 to 6,500). The greatest concentrations of dissolved aluminum in Newport Basin were measured in AMD from several relatively small (flows from 0.02 to 0.89 ft^3/s) AMD discharges entering South Branch Basin between Wananie and Sheatown (dissolved aluminum concentration from 5,600 to 6,900 $\mu g/L$).

The chemistry of stream water after mixing with AMD was variable because of a number of factors. Extended residence time during low flows allowed for more complete hydrolysis of iron and greater acidity production. This process, all or in part, may explain the unusually acidic conditions (minimum pH was 3.2) observed in the commonly near-neutral (median pH was 6.8) water of Nanticoke Creek about 0.5 mi downstream of the Truesdale Mine Discharge. Other co-occurring influences may explain pH variability observed near the mouth of South Branch (pH range was 3.6 to 6.9). The presence of *Escherichia coli* bacteria, along with corresponding ammonia data, indicated intermittent sewage inflows contaminated South Branch and other reaches. Alkalinity contributed by the sewage buffered acidity in the lower reach South Branch and explained the bimodal pH distribution; water had near-neutral pH (pH greater than 6.0) when sewage was abundant and acidic pH (less than 4.3) when it was not.

Given data on the variability of AMD quality and flow, various passive- and limited active-treatment strategies were identified that could ameliorate most AMD in the study area. AMD in other mined areas with chemistry and flow similar to the discharges sampled for this study has been treated with passive strategies that may include amendment of influent chemistry and almost always include retention in aerobic wetlands. For this study, aerobic-wetland sizes were computed assuming iron removal rates from 5 to 20 (g/m^2)/d. Although a portion of AMD from the Truesdale Mine Discharge is presently treated in aerobic wetlands covering less than 2 acres, additional wetland

area may be needed when the AMD quality is net acidic. In similar situations with iron-loading rates approaching 326 kg/d, aerobic wetlands of 4 to 16 acres combined with an alkalinity source have been used for passive treatment of the AMD. For the Susquehanna Number 7 Mine, a wetland of 12 to 49 acres could be considered to remove dissolved iron, without supplemental alkalinity, provided that pH is maintained near neutrality. Aeration to drive off CO_2 has been used to accelerate the oxidation rate and removal of iron at other net-alkaline discharges similar to the Susquehanna Number 7 Mine. Accelerating iron removal by aeration would decrease retention time and the corresponding wetland area needed for treatment. AMD sources with large flow rates, low pH, and elevated concentrations of dissolved metals, such as the mine-pit overflow, may warrant active treatment. For example, an efficient alkalinity-producing system, such as lime dosing, followed by a pond or wetland of approximately 3 to 13 acres has been used to neutralize the AMD and remove the dissolved iron in similar situations.

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