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Assessment of Characteristics and Remedial Alternatives for Abandoned Mine Drainage: Case Study at Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Cambria County, Pennsylvania, 2004

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Conversion Factors

Inch/Pound to SI

Multiply	By	To obtain
Length		
inch (in.)	2.54	centimeter (cm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
Area		
square mile (mi^2)	2.590	square kilometer (km^2)
Volume		
cubic foot (ft^3)	0.02832	cubic meter (m^3)
Flow rate		
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)

Abstract

This report describes field, laboratory, and computational methods that could be used to assess remedial strategies for abandoned mine drainage (AMD). During April-June, 2004, the assessment process was applied to AMD from bituminous coal deposits at a test site in the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site (ALPO-SBTU) in Cambria County, Pennsylvania. The purpose of this study was (1) to characterize the AMD quantity and quality within the ALPO-SBTU test site; (2) to evaluate the efficacy of limestone or steel slag for neutralization of the AMD on the basis of reaction-rate measurements; and (3) to identify possible alternatives for passive or active treatment of the AMD. The data from this case study ultimately will be used by the National Park Service (NPS) to develop a site remediation plan. The approach used in this study could be applicable at other sites subject to drainage from abandoned coal or metal mines.

During April 2004, AMD from 9 sources (sites1, 1Fe, Fe, 2, 3, 3B, 5, 6, and 7) at the ALPO-SBTU test site had a combined flow rate of 1,420 gallons per minute (gal/min) and flow-weighted averages for pH of 3.3, net acidity of 55 milligrams per liter (mg/L) as $CaCO_3$, and concentrations of dissolved sulfate, aluminum, iron, and manganese of 694 mg/L, 4.4 mg/L, 0.74 mg/L, and 1.2 mg/L, respectively. These pH, net acidity, sulfate, and aluminum values exceed effluent criteria for active mines in Pennsylvania.

During April-June 2004, limestone and steel slag that were locally available were tested in the laboratory for their composition, approximate surface area, and potential to neutralize samples of the AMD. Although the substrates had a similar particle-size distribution and identical calcium content (43 percent as calcium oxide), the limestone was composed of crystalline carbonates and the slag was composed of silicate glass and minerals. After a minimum of 8 hours contact between the AMD and limestone or steel slag in closed containers (cubitainers), near-neutral effluent was produced. With prolonged contact between the AMD and limestone or steel slag, the concentrations of iron, aluminum, and most dissolved trace elements in effluent from the cubitainers declined while pH was maintained greater than 6.0 and less than 9.0. The cubitainer testing demonstrated (1) lower alkalinity production but higher pH of AMD treated with steel slag compared to limestone, and (2) predictable relations between the effluent quality, detention time, and corresponding flow rate and bulk volume for a bed of crushed limestone or steel slag in an AMD passive-treatment system.

The process for evaluating AMD remedial strategies at the ALPO-SBTU test site involved the computation and ranking of the metal loadings during April 2004 for each of the AMD sources and a comparison of the data on AMD flow and chemistry (alkalinity, acidity, dissolved oxygen, ferric iron, aluminum) with published criteria for selection of passive-treatment technology. Although neutralization

of the AMD by reaction with limestone was demonstrated with cubitainer tests, an anoxic limestone drain (ALD) was indicated as inappropriate for any AMD source at the test site because all had excessive concentrations of dissolved oxygen and (or) aluminum. One passive-treatment scenario that was identified for the individual or combined AMD sources involved an open limestone channel (OLC) to collect the AMD source(s), a vertical flow compost wetland (VFCW) to add alkalinity, and an aerobic wetland to facilitate iron and manganese oxidation and retention of precipitated solids. Innovative passive-system designs that direct flow upward through submerged layers of limestone and/or steel slag and that incorporate siphons for automatic flushing of solids to a pond also may warrant consideration. Alternatively, an active-treatment system with a hydraulic-powered lime doser could be employed instead of the VFCW or upflow system. Now, given these data on AMD flow and chemistry and identified remedial technologies, a resource manager can use a publicly available computer program such as "AMDTreat" to evaluate the potential sizes and costs of various remedial alternatives.

Introduction

Abandoned mine drainage (AMD) affects the quality and potential uses of water supplies in coal and metal mining regions worldwide (Herlihy *et al*., 1990; Nordstrom, 2000). AMD ranges widely in quality from mildly alkaline to strongly acidic and corrosive, with dissolved solids ranging from about 200 to 10,000 mg/L (Hyman and Watzlaf, 1997; Rose and Cravotta, 1998; Nordstrom and Alpers, 1999). AMD characteristically has elevated concentrations of dissolved sulfate, iron, and other metals. Dissolved metals and other constituents in AMD can be toxic to aquatic organisms and ultimately can precipitate forming ochreous encrustations that degrade the aquatic habitat (Winland *et al*., 1991; Bigham and Nordstrom, 2000).

The pH and concentrations and loadings of alkalinity, acidity, and metals such as iron (Fe), aluminum (Al), and manganese (Mn) in mine effluent and receiving water bodies commonly are measured to identify potential for environmental effects (Commonwealth of Pennsylvania, 1998a, 1998b, 2002; U.S. Environmental Protection Agency, 2002a, 2002b). These parameters also are measured to identify appropriate treatment methods to remove the metals and maintain neutral pH (Hedin *et al*., 1994; Skousen *et al*., 1998). The pH of AMD is an important measure for evaluating chemical equilibrium, corrosiveness, and aquatic toxicity. The severity of toxicity or corrosion tends to be greater under low-pH conditions than under near-neutral conditions. For example, Al is soluble at low pH, and compared to Fe and Mn, relatively low concentrations of dissolved Al can be toxic (Elder, 1988; Bigham and Nordstrom, 2000). Accordingly, the U.S. Environmental Protection Agency (2000, 2002a, 2002b) recommends pH 6.5 to 8.5 for public drinking supplies and pH 6.5 to 9.0 for protection of freshwater aquatic life. Furthermore, the

Commonwealth of Pennsylvania (1998a, 1998b, 2002) stipulates that effluent discharged from active mines must have pH 6.0 to 9.0 *and* alkalinity greater than acidity.

Recently, resource managers have gained access to the publicly available AMDTreat computer program for evaluation of the approximate construction and maintenance costs of active or passive systems for treatment of AMD (U.S. Office of Surface Mining Reclamation and Enforcement, 2002). Only the AMD flow rate and concentrations of acidity, alkalinity, iron, manganese, and aluminum are required as input data for this program. However, inappropriate comparisons of remedial strategies and poor decisions can result because the AMDTreat program does not consider if the selected AMD treatment technology meets recommended criteria for implementation.

Different alternatives for treatment of AMD could be appropriate depending on the volume of the mine discharge, its alkalinity and acidity balance, its concentrations of dissolved oxygen and metals, and the available resources for construction and maintenance of a treatment system (Hedin *et al*., 1994; Skousen *et al*., 1998). If the effluent is "net alkaline," the alkalinity exceeds the acidity and the pH will remain near neutral after complete oxidation of the effluent. In this case, systems that facilitate aeration of the effluent and retention of precipitated solids are indicated. On the other hand, if the effluent is "net acidic," the acidity exceeds the alkalinity and the pH will decline to acidic values after complete oxidation and precipitation of the dissolved metals. In this case, systems that add alkalinity and that maintain or increase pH are indicated.

Selection of a Test Site

The Allegheny Portage Railroad in Cambria County, Pennsylvania, was constructed during 1831-1834 as a 36-mile-long inclined plane railroad over the Allegheny Mountains to connect canal segments along a 394-mile transportation route between Philadelphia and Pittsburgh, Pa. (Sellards & Grigg, Inc., 1991). Hailed as an engineering marvel in 1834, the canal and railroad reduced travel time between Philadelphia and Pittsburgh from three weeks by wagon to only four days (VisitPA.com, 2005). The 901-foot long Staple Bend Tunnel, excavated at the head of Plane 1 through a promontory that formed a bend in the Little Conemaugh River, was the first railroad tunnel to operate in the United States (Sellards & Grigg, Inc., 1991). Designated a National Historic Site in 1964 and acquired by the U.S. Department of Interior in 1991 from Bethlehem Steel Corporation, the present site is managed by the National Park Service (NPS) (Sellards & Grigg, Inc., 1991; National Park Service, 2005). The site covers 1,249 acres and includes various historical attractions plus hiking and biking trails. The site also includes remnants of abandoned coal mines and abuts the Cambria steel slag dump that postdate the 1833-1852 period of historical significance (Sellards & Grigg, Inc., 1991).

Historical mining of bituminous coal in the vicinity of the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site (ALPO-SBTU) has caused widespread contamination of water resources within the ALPO-SBTU boundaries and surrounding watersheds (Pennsylvania Department of Environmental Protection, 2001; Kaktins and Carney, 2002). The ALPO-SBTU is near the eastern limit of the Bituminous Coalfield and the northern Appalachian Plateau (Edmunds, 1999). AMD from underground and surface mines along the eastern perimeter of the ALPO-SBTU sustains perennial flows in several unnamed tributaries that cross the ALPO-SBTU and ultimately discharge into the Little Conemaugh River (fig. 1). The AMD within the park was cited as a deficiency in a 2002 NPS Environmental Audit (Alan C. Ellsworth, National Park Service, 2004, written commun.).

Data collected within the ALPO-SBTU test site by Kaktins and Carney (2002) for drought conditions in 2000-2001 indicated ranges for AMD flow rates, pH, acidity, and metals loading that could feasibly be remediated using passive-treatment technology (Hedin *et al*., 1994; Skousen *et al*., 1998). However, the flow rates and associated contaminant-loading rates for drought conditions could underestimate the average or long-term conditions and, consequently, result in an incorrect basis for design of remediation. Nevertheless, the site characteristics are suitable for testing the assessment process for remedial treatment of AMD.

Treatment systems require a neutralization media that is economical to use. An important factor in selection of this site for study is that limestone and steel slag are locally available and potentially useful for mine-drainage treatment. Calcium-bearing compounds in the limestone and steel slag can neutralize AMD, increasing pH and alkalinity (Ziemkiewicz and Skousen, 1998; Simmons *et al*., 2002a, 2002b; Cravotta, 2003). However, equal amounts of limestone or slag could have different neutralizing capacities and rates of reaction because limestone predominantly is composed of crystalline carbonates and slag predominantly is composed of silicate glass. Furthermore, the pH, dissolved carbon dioxide (CO_2), and other solutes in AMD could affect the rates of reaction in a treatment system (Cravotta and Watzlaf, 2002; Cravotta, 2003). Hence, investigation of these factors is warranted to determine potential effectiveness of using limestone or steel slag for treatment of the AMD at the ALPO-SBTU test site.

Purpose and Scope

This report outlines a process to assess remedial alternatives for treatment of AMD. The ALPO-SBTU test site in Cambria County, Pennsylvania, is used to illustrate how the process can be applied. The report summarizes (1) the quantity and quality of the AMD within the ALPO-SBTU test site during April 2004, (2) the efficacy of limestone and steel slag for treatment of the AMD on the basis of laboratory reaction-rate measurements, and (3) possible remediation strategies for passive and active treatment of the AMD on

the basis of the April 2004 water-quality data and available criteria for identification of passive-treatment technologies. Synoptic field surveys and laboratory studies were conducted by the U.S. Geological Survey (USGS) in cooperation with the NPS in April 2004, to acquire data for nondrought, high base-flow conditions and to evaluate limestone and steel slag for treatment of the AMD at the ALPO-SBTU test site. The approach used in this study could be applicable at other sites subject to drainage from abandoned coal or metal mines.

Data Collection and Evaluation

Data were collected by the USGS on the flow rates and composition of AMD and associated pond water and sediment, and the rate of reaction between AMD and limestone or steel slag. Methods used to collect and evaluate these data are summarized below.

Field Methods

Synoptic surveys were conducted by the USGS during high base-flow conditions on April 7 and 27, 2004, to characterize the inorganic chemistry of all known AMD, including 12 sites previously sampled by Kaktins and Carney (2002), 1 previously unsampled discharge site, and 6 sites on two previously unsampled ponds downstream from AMD sources within the ALPO-SBTU boundaries (table 1). At each sample site, temperature, pH, specific conductance (SC), dissolved oxygen (DO), and redox potential (Eh) were measured using a multiparameter, submersible sonde calibrated in the field in accordance with standard field methods (U.S. Geological Survey, 1997 to present). Field pH and Eh were determined using a combination Pt and Ag/AgCl electrode with a pH sensor. The electrode was calibrated in pH 2.0, 4.0, and 7.0 buffer solutions and in ZoBell solution (Wood, 1976, p. 18-22; U.S. Geological Survey, 1997 to present). Values for Eh were corrected to 25 °C relative to the standard hydrogen electrode in accordance with methods of Nordstrom (1977). Where weirs were intact, the flow rate was estimated on the basis of the water depth flowing through the weir notch and the appropriate weir equation given the notch geometry (Rantz *et al*., 1982b; Kaktins and Carney, 2002). Where weirs were absent or damaged, the flow rate was measured using a wading rod and pygmy current meter or a bucket and stop watch (Rantz *et al*., 1982a). Unfiltered and filtered (0.45-micrometer (μm) pore size) samples of water from each AMD source and pond sampling point were processed in the field, transferred to polyethylene bottles, preserved as appropriate, and transported on ice to the laboratory.

On April 27, 2004, a canoe was used to access each pond for sampling of water and bottom sediment. The temperature, pH, and SC near the top and bottom of the water column were measured while sampling locations were recorded with a global positioning system (GPS) and electronic depth indicator. Near the

upstream (inflow) and downstream (outflow) ends of each pond, a clamshell dredge device was suspended from the canoe to collect bottom sediments for analysis of chemistry and mineralogy. The wet sediment was transported on ice in a sealed polyethylene bag to the laboratory.

Laboratory Methods

The alkalinity and "hot" acidity of the unfiltered water samples were titrated using standard methods to fixed endpoint pH of 4.5 and 8.3, respectively, within 48 hr of sampling at the USGS New Cumberland laboratory (American Public Health Association, 1998a, b). Concentrations of sulfate, chloride, and nitrate in filtered, unpreserved subsamples were analyzed by ion chromatography (IC), and concentrations of dissolved metals in filtered, acidified subsamples were analyzed by inductively coupled plasma emission mass spectrometry (ICP-MS) at the Actlabs laboratory in Toronto, Ontario (Crock *et al*., 1999).

The pond-bottom sediments and samples of limestone and steel slag used for cubitainer testing (described below) were air dried and pulverized in the laboratory prior to analysis of mineralogy and chemistry. The mineralogy was analyzed by X-ray diffraction (XRD) at the USGS Minerals Resources laboratory in Reston, Virginia (Whittig and Allardice, 1986; Taggart, 2002). The XRD patterns were collected on a Scintag theta-theta diffractometer using copper radiation over the range of 3° to 70° with a 0.02° step size and a count time of 1 second per step (Nadine Piatak, U.S. Geological Survey, written commun., 2004). Major, minor, and trace elements in a 1-gram subsample of the dried sediment were analyzed by inductively coupled plasma atomic emission spectrometry (ICP-AES) and ICP-MS after lithium metaborate-tetraborate fusion and decomposition with a mixture of hydrochloric, nitric, perchloric, and hydrofluoric acids (Lim and Jackson, 1982; Crock *et al*., 1999) at the Actlabs laboratory in Ontario, Canada.

"Cubitainer" tests were conducted during May 3-17, 2004, in the USGS New Cumberland laboratory to evaluate quantities of limestone or steel slag needed for AMD treatment using methods of Cravotta (2003) and Cravotta and others (2004). Following the synoptic survey on April 27, water samples from sites Fe and 1 were collected into separate 5-gallon polyethylene containers, sealed without headspace, immediately transferred on ice to the laboratory, and then refrigerated. In the laboratory, the untreated water from each site was transferred to a 1-gallon collapsible polyethylene container (cubitainer) containing 2 kilograms (kg) of limestone or steel slag particles that had been sieved to nominal diameters ranging from 0.5 to 1.5 inches (1.3 to 3.8 centimeters (cm)). AMD from site 1 was added to one cubitainer filled with limestone and a second filled with steel slag, and AMD from site Fe was added to a third cubitainer filled with limestone and a fourth filled with steel slag. Effluent samples from each of the four cubitainers were collected at progressively longer detention times, starting at 0.5-hour (hr) intervals, over 2

weeks total elapsed time. Each effluent sample was withdrawn using a 120-milliliter (mL) syringe, and a 60-mL aliquot was pushed through a 0.45-µm pore-size nylon filter. The first 10-mL filtrate of the 60-mL aliquot was discarded and the remaining 50-mL filtrate was analyzed immediately for pH, alkalinity, and calcium concentrations by electrometric and colorimetric titration methods (American Public Health Association, 1992a, 1998b). A subset of the effluent samples collected after 1 hr, 8 hr, and 336 hr elapsed detention time also was analyzed for trace metals by ICP-MS to assess potential for contaminants to be derived from, or removed by, reaction with the limestone and steel-slag substrates.

The limestone and steel slag used for the laboratory rate tests were obtained from commercial sources near the ALPO-SBTU test site with the expectation that the same materials could be used for construction of a treatment system. The limestone, with a reported $CaCO_3$ content of 83 weight percent, was obtained from the Ashcom Quarry, near Everett, Bedford County, through New Enterprise Stone and Lime Co., Inc., of New Enterprise, Pa. According to O'Neill (1964), this quarry is in the undivided Middle Ordovician Coburn through Loysburg Formations. The Johnstown limestone that can be present locally with coal-bearing strata of the Pennsylvanian System (Brady *et al*., 1998) is not quarried because it is relatively thin and impure (Keith B. C. Brady, Pennsylvania Department of Environmental Protection, oral commun., 2004). The steel slag, originally produced as a blast-furnace byproduct, was obtained through Rollock Inc., of Johnstown, Pa., from stockpiles near the ALPO-SBTU test site (fig. 1). Commercial uses of the stockpiled slag include railroad ballast, slope protection, anti-skid material, roofing granules, embankments, and fills (Simmons *et al*., 2002a). The slag had been sieved and cleaned commercially by magnetic separation to eliminate stray scraps of iron and steel.

Data-Evaluation and Computational Methods

Various water- and sediment-quality guidelines are relevant for evaluating the quality of the AMD and the pond water and sediment at the ALPO-SBTU test site. Effluent from a coal mine that was 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 milligrams per liter (mg/L), 5.0 mg/L, or 0.75 mg/L, respectively (Commonwealth of Pennsylvania, 1998a, b). 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, b). 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 watersheds (Commonwealth of Pennsylvania, 2001a; Pennsylvania Department of Environmental

Protection, 2002, 2004). 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) or 87°F (30.6°C) for CWF and WWF, respectively;
- dissolved oxygen concentration greater than 5.0 mg/L for CWF and 4.0 mg/L for WWF;
- alkalinity not less than 20 mg/L as $CaCO_3$, except where natural conditions are less;
- pH not less than 6.0 or greater than 9.0;
- total iron concentration not to exceed 1.5 mg/L as a 30-day average;
- dissolved iron concentration not to exceed 0.3 mg/L;
- total manganese concentration not to exceed 1.0 mg/L; and
- total aluminum concentration not to exceed 0.75 mg/L.

TMDLs for the Little Conemaugh River are proposed for development during 2007-2009 (Pennsylvania Department of Environmental Protection, 2004). Additional water-quality criteria established by the U.S. Environmental Protection Agency (2004) 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 *et al*., 2000), but these sediment-quality guidelines have not been adopted by regulatory authorities in the United States.

Following methods of Cravotta and Kirby (2004), the net-acidity concentrations for the AMD and pond-water samples were computed as the difference between the computed acidity based on the pH and the concentrations of dissolved aluminum, iron, and manganese and the measured alkalinity. Generally, with aeration and aging of AMD, the pH can decrease because of oxidation and hydrolysis of dissolved ferrous (Fe^{II}) and manganous (Mn^{II}) species and the consequent precipitation of solid phases.

$$Fe^{2+} + 0.25\ O_2 + 2.5\ H_2O \rightarrow Fe(OH)_3\ (s) + 2\ H^+ \quad (1)$$

$$Mn^{2+} + 0.5\ O_2 + H_2O \rightarrow MnO_2\ (s) + 2\ H^+ \quad (2)$$

Precipitation of $Fe(OH)_3$, MnO_2, and associated solids, such as FeOOH and $Al(OH)_3$, will consume some or all available alkalinity. The net acidity (= computed acidity - measured alkalinity) indicates the potential for the pH to decrease to acidic values because of the oxidation and hydrolysis of dissolved metals (Cravotta and Kirby, 2004). The computed net acidity of AMD is equivalent to the measured hot acidity

(Cravotta and Kirby, 2004). If the net acidity is negative (< 0), the effluent ultimately will be near neutral (pH = 6 to 8) and some alkalinity will remain after complete oxidation and hydrolysis of the dissolved metals. However, if the net acidity is positive (> 0), the effluent ultimately will be acidic (pH < 4.5) and an additional source of alkalinity would be needed to neutralize the excess acidity.

The maximum measured flow and maximum concentrations of net acidity, dissolved oxygen, and metals for the April 2004 high base-flow samples were used to estimate relative contributions of AMD pollutants and to identify possible remedial alternatives for each of the AMD sources at the ALPO-SBTU test site on the basis of published criteria for selection of passive-treatment technology (fig. 2). The net-acidity and metals loading for each AMD source were computed as the product of flow rate and the relevant constituent concentrations to indicate the environmental significance of the sources. These loading computations also could be useful to evaluate sizing of possible AMD treatment systems. For example, the iron-loading rate can be used to estimate the size of an aerobic wetland (Hedin and others, 1994; Pennsylvania Department of Environmental Protection, 1999), and the annual net-acidity loading (net acidity > 0) can be used to indicate the annual quantities of limestone required for a passive neutralization within an anoxic limestone drain (ALD) or vertical flow compost wetland (VFCW) (Watzlaf and others, 2000; Rose, 2004) or for active treatment with hydrated lime (U.S. Office of Surface Mining Reclamation and Enforcement, 2002). Nevertheless, such sizing estimates based on loading rates ignore characteristics of the water quality and available substrates that can affect reaction rates and, consequently, treatment-system performance.

Following methods of Cravotta and others (2004), the titration data for the cubitainer tests were used to derive first-order and second-order rate equations to estimate the rates of substrate dissolution and of increases in alkalinity and calcium concentrations as a function of the detention time within a limestone or steel-slag bed used for a passive-treatment system. By combining the cubitainer rate estimates with estimates for the initial mass of limestone or steel slag, the porosity of the bed (assumed to be 45 percent), and the reported AMD discharge rate, exponential decay models were obtained. These models indicate possible long-term trends, on a decadal scale, for changes in mass of limestone or slag, detention time, and corresponding concentrations of alkalinity of the effluent with age of the treatment system. Calcite saturation index (SI) and partial pressure of carbon dioxide (Pco_2) were computed using measured values for temperature, SC, pH, alkalinity, and calcium (Langmuir, 1997; American Public Health Association, 1992b); van't Hoff temperature-corrected equilibrium constants from Ball and Nordstrom (1991); and Debye-Huckel activity coefficients on the basis of ionic strength estimated from SC (Langmuir, 1997). The concentration of dissolved Fe^{II} was computed on the basis of the measured Eh, temperature, and ionic strength of fresh samples (Nordstrom, 1977, 2000).

Characteristics of Abandoned Mine Drainage

Data on the flow rates and composition of AMD, pond water and sediment, and the effluent and solids from cubitainers were used to assess the characteristics, possible ecological effects, and to identify possible alternatives for passive remediation of the AMD within the ALPO-SBTU test site. These data are summarized and evaluated below and, ultimately, will be used by resources managers of the NPS to develop a site remediation plan.

Abandoned Mine Drainage Flow Rates and Chemistry

Water-quality data for the field and laboratory analyses of the AMD and pond samples collected April 7 and 27, 2004, are summarized in tables 2 and 3 and figures 3 and 4. A total of 13 samples were collected from known AMD sources and selected downstream locations on April 7, 2004. A subset of these sites was resampled on April 27, 2004, with additional sampling conducted within the two ponds near the southwest boundary of the ALPO-SBTU test site. Data collected on April 7 (tables 2 and 3) and previously by Kaktins and Carney (2002) indicated similar chemistry for upstream and downstream pairs of samples, but larger flow rates at the downstream sites. Thus, sites U, 3A, and 4 were excluded from the second set of samples on April 27 because these were upstream of sites 1, 3B, and 5, respectively. Site 2 was excluded from the second set of samples because it is not a separate flow, but represents combined seepage from sites 1Fe, Fe, and other diffuse discharges associated with an iron mound; samples from sites 1Fe and Fe were presumed representative of the chemistry of this seepage. Nevertheless, to account for the combined seepage flow volumes, data for site 2 were included in various computations.

On April 7 and 27, 2004, the AMD flow rates ranged from 0.4 gal/min to 498 gal/min (table 2, fig. 3A), with the largest flows, exceeding 350 gal/min, at sites 1, 3, and 5. Previously reported maximum flows at sites 1, 3, and 5 were 232, 73, and 105 gal/min, respectively (Kaktins and Carney, 2002). All the known AMD sources were flowing in April 2004; however, previously, some of the small AMD sites had been reported as intermittent (Kaktins and Carney, 2002).

With the exception of site 8, the AMD samples for April 2004 were net acidic and had elevated concentrations of dissolved sulfate, silica, and metals compared to background. Site 8 had net-alkaline water quality with near-neutral pH, moderate concentrations of sulfate (from 317 to 351 mg/L), and low concentrations of dissolved metals, consistent with background (table 2, figs. 3 and 4). The pH ranged from 2.3 at site 2 to 7.8 at site 8 (table 2, fig. 3B). Net acidity ranged from -45 mg/L as $CaCO_3$ at site 8 to 322 mg/L as $CaCO_3$ at site 2. Concentrations of sulfate ranged from 317 to 1,090 mg/L and silica ranged from 5.8 to 34.7 mg/L at site 8 and site Fe, respectively. Except for sites Fe, 1Fe, and 2, the concentrations

of dissolved aluminum exceeded those of iron and manganese (table 2). Concentrations of dissolved aluminum ranged from <0.02 mg/L at site 8 to 15.9 mg/L at site 6. Concentrations of dissolved iron ranged from <0.1 mg/L at site 8 to 76.4 mg/L at site Fe. Concentrations of dissolved manganese ranged from <0.01 mg/L at site 8 to 4.95 mg/L at site 1Fe.

Although concentrations of manganese for all the AMD sources were less than the 5-mg/L effluent limit for active mines in Pennsylvania (fig. 3G), concentrations of aluminum for most of the AMD sources exceeded the 0.75-mg/L effluent limit (fig. 3F). Additionally, AMD at sites Fe, 1Fe, and 2 had concentrations of iron that exceeded the 7.0-mg/L effluent limit (fig. 3E). Furthermore, all the AMD samples, including site 8, had concentrations of sulfate that exceeded the 250-mg/L limit that applies for mine discharges that could affect public water supplies (fig. 3D).

The quality of all the AMD sources combined was approximated by the computation of the flow-weighted average concentrations of constituents and the sum of flow and loading rates. The cumulative flow rate for the combined AMD sources was 1,420 gal/min on April 7 and 1,231 gal/min on April 27. This calculation excludes site 8 (fig. 1), because it is isolated from other sources and is not representative of AMD, and sites U, 3A, and 4, because these sites are redundant with downstream sites. The relevant sources had a flow-weighted average pH of 3.3 (determined on the basis of flow-weighted hydrogen-ion concentration) and net acidity of 55 mg/L as $CaCO_3$. The flow-weighted average concentrations of sulfate, silica, aluminum, iron, and manganese were 694 mg/L, 17.2 mg/L, 4.4 mg/L, 0.74 mg/L, and 1.2 mg/L, respectively. On the basis of these values, the pH, net acidity, sulfate, and aluminum concentrations would exceed relevant effluent criteria for active mines (Commonwealth of Pennsylvania, 1998a, b) (fig. 3).

The sum of iron, aluminum, and manganese loading for the combined AMD sources was 19.7 and 16.6 tons per year (ton/yr) on April 7 and 27, 2004, respectively (table 2). Site 1 had the largest metals loading rate, discharging an average of 6.5 ton/yr of iron, aluminum, and manganese (table 2, fig. 3H). Sites 3, 5, 6, and 7 each discharged approximately half that load, largely from dissolved aluminum (table 2, fig. 3H).

In addition to aluminum, iron, and manganese, dissolved trace metals were present in the AMD at the ALPO-SBTU test site (table 3, fig. 4). Concentrations of arsenic, cadmium, chromium, copper, lead, and zinc in the AMD did not exceed relevant U.S. Environmental Protection Agency (2002b) drinking-water standards (fig. 4); however, drinking-water standards are not available for cobalt, nickel, and various other metals. Nevertheless, concentrations of cobalt, nickel, and zinc in the AMD samples (figs. 4C, 4G, 4H) consistently exceeded continuous concentration criteria for protection of freshwater aquatic life (CCCF) (U.S. Environmental Protection Agency, 2002a; Commonwealth of Pennsylvania, 2001b). Additionally,

chromium, copper, and lead concentrations in several of these samples exceeded relevant CCCF values (figs. 4D, 4E, 4F).

The water-quality samples collected April 27 had higher pH and lower concentrations of net acidity, sulfate, and most dissolved metals compared to samples collected April 7 (tables 2 and 3, figs. 3 and 4). Furthermore, the samples collected in April 2004 generally had lower concentrations of net acidity and dissolved metals than the samples reported by Kaktins and Carney (2002) for lower flow conditions than the present study. Hence, the data from Kaktins and Carney (2002) along with the data for April 2004 indicate a general decline in the concentrations of acidity and metals associated with increased flow rate. Dilution of mineralized ground water with meteoric water (rainwater, snowmelt) could explain the relatively low concentrations of dissolved sulfate and metals for the high base-flow samples. Kaktins and Carney (2002) noted that acidity concentrations for the AMD sources tended to be larger during fall low-flow conditions compared to spring high-flow conditions during their study, but acidity loading was larger in the spring. If dilution or evaporation was the sole process causing changes in flow rate and associated chemical concentrations, the chemical load would not change. The increase in acidity loading associated with higher flow conditions in the spring during 2001 and the current study implies that the meteoric water is a dilute acidic solution that adds some acidity, sulfate, and metals, likely due to atmospheric aerosols (acid rain) and dissolution of pyrite-oxidation products along the recharge pathway (Cravotta, 1994, 2000).

Pond-Water Volume and Chemistry

The two ponds near the southern limit of the ALPO-SBTU test site are bounded on the east bank by the hillside leading to the historical rail trail and abandoned mines and on the west by the active railroad and the Little Conemaugh River (fig. 1). Although water from the upper pond (pond #1) is presumed to flow to the lower pond (pond #2), no direct spillover or surface flow was apparent from either pond in April 2004. Seepage to pond #2 is likely to take place through the earthen berm separating the two ponds. Likewise, water is likely to seep from both ponds to the Little Conemaugh River through the adjacent railroad bed. Elevation surveys were not conducted to determine precise volumes or capacity. However, based on the GPS coordinates and water-column depth readings completed on April 27, 2004, the upper pond has a surface area of approximately 1.5 acres with a volume of approximately 4.47 acre-feet (acre-ft) based on its length of 860 feet (ft), average width of 75 ft, and average depth of approximately 3 ft. The lower pond has a surface area of approximately 0.9 acres with a volume of 4.29 acre-ft, based on its length of 440 ft, average width of 85 ft, and average depth of approximately 5 ft.

The chemistry of water from the upper pond indicated it originated as a mixture of AMD from various sources (sites 3, 3B, 5, 6, 7, and other unidentified seeps) plus smaller quantities from non-AMD or

alkaline sources (site 8 and other associated seeps). Samples from the upper pond (pond #1 outflow) on April 7 and 27, 2004, had pH of 3.8 and 3.5, net acidity of 38 and 41 mg/L as $CaCO_3$, and concentrations of dissolved sulfate of 356 and 358 mg/L, silica of 15.7 mg/L, aluminum of 5.07 and 3.94 mg/L, iron of 0.25 and 0.41 mg/L, and manganese of 0.84 and 0.91 mg/L, respectively (table 2). In comparison, the main tributary to the upper pond (pond #1 inflow) had flow rates of 260 and 324 gal/min with corresponding pH of 3.2 and 3.6, net acidity of 90 and 60 mg/L as $CaCO_3$, and concentrations of dissolved sulfate of 536 and 499 mg/L, silica of 22.7 and 24.0 mg/L, aluminum of 9.25 and 7.62 mg/L, iron of 0.45 and 0.60 mg/L, and manganese of 1.48 and 1.46 mg/L, respectively (table 2). Combining data for April 7 and 27, 2004 (table 2), the AMD sampled at sites 3, 3B, 5, 6, and 7 had a combined average flow rate of 955 gal/min (average of combined flow rates of 1,042 and 868 gal/min for April 7 and 27, respectively) with flow-weighted average values for pH of 3.7, net acidity of 32 mg/L as $CaCO_3$, and concentrations of sulfate, silica, aluminum, iron, and manganese of 706 mg/L, 15.8 mg/L, 3.87 mg/L, 0.35 mg/L, and 0.93 mg/L, respectively. These flow-weighted average values do not change substantially if data for site 8 are included, because of its small flow rate, but would tend to approach measured values in the pond if proportionally greater volumes of such non-AMD were combined with the AMD. Thus, although the pond water was similar in composition to that computed for combined AMD sources, the computed volume of the combined AMD sources was three times larger than the measured inflow to pond #1. The relatively small inflow volume entering pond #1 compared to the combined AMD volume indicates a large fraction of the upstream AMD at sites 3, 3B, 5, 6, and 7 exits the ALPO-SBTU test site boundaries, possibly as seepage through the railroad bed, between the AMD sources and pond #1. Furthermore, additional non-AMD sources that were not sampled or quantified flow to the ponds as surface runoff and seepage. Although contributions from the non-AMD sources tend to have a diluting effect on sulfate and metals concentrations, they have little effect on the pH of the ponds.

In April 2004, the chemistry of water from the lower pond (pond #2 inflow and outflow) was similar to that for the outflow of pond #1 (tables 2 and 3). Except for arsenic and chromium, the outflow samples from pond #1 and pond #2 had lower concentrations of net acidity, sulfate, and dissolved metals than the inflow to pond #1 (tables 2 and 3). Lower concentrations of sulfate and dissolved metals in the outflow from pond #1 and pond #2 compared to the inflow to pond #1 and the average for sites 3, 3B, 5, 6, and 7 indicate that attenuation of these constituents by dilution and precipitation is likely to occur within the ponds. For example, mixing of the alkaline water from site 8 with acidic water from sites 3, 3B, 5, 6, and 7 results in intermediate net acidity, sulfate, and metals concentrations, with potential for precipitation of iron and aluminum hydroxide, hydroxysulfate, and (or) silicate minerals. Nevertheless, dissolved cobalt, chromium, nickel, and zinc in both ponds were consistently greater than the relevant CCCF values for

protection of aquatic life (table 3). Concentrations of copper and lead were similar to the CCCF values, and arsenic and cadmium were less than the CCCF values (table 3).

Mineralogy and Chemistry of Pond Sediments

Aluminum-rich precipitate with a light-brownish color accumulated on the bottom of the upper pond where the metal-laden AMD (sites 3, 3B, 5, 6, 7, and other unidentified seeps) mixed with alkaline water from various sources (site 8 and other associated seeps). The precipitate, which was thickest near the inflow to pond #1, was composed predominantly of aluminum- and silica-bearing compounds, such as quartz (SiO_2), muscovite ($KAl_3Si_3O_{10}(OH)_2$), kaolinite ($Al_2Si_2O_5(OH)_4$), chlorite ($Mg_5Al_2Si_3O_{10}(OH)_8$), and poorly crystalline or amorphous aluminum-hydroxysulfate minerals (table 4). The same minerals were present in sediment samples near the outflow of pond #1 and the inflow of pond #2. Although XRD patterns did not reveal iron minerals, trace quantities of ferrihydrite ($Fe(OH)_3$; nominally $Fe_5HO_8{\cdot}4H_2O$), goethite (α–FeOOH), schwertmannite ($Fe_8O_8(OH)_6SO_4$), and other Fe^{III} minerals (Winland *et al*., 1991; Bigham *et al*., 1996) likely imparted the rusty brownish colors to the sediments.

The bottom sediments sampled from both ponds at the ALPO-SBTU test site were enriched in various metals (tables 4 and 5). The pond sediments had as much as three times higher concentrations of aluminum, arsenic, and chromium, but equivalent or lower concentrations of iron, copper, nickel, lead, and zinc compared to median concentrations for streambed sediments across the conterminous United States that were reported by Rice (1999) for aluminum (Al_2O_3 = 12.1 weight percent (wt%)), iron (Fe_2O_3 = 5.0 wt%), arsenic (As = 6.3 parts per million (ppm)), chromium (Cr = 64 ppm), copper (Cu = 27 ppm), nickel (Ni = 27 ppm), lead (Pb = 27 ppm), and zinc (Zn = 110 ppm). Of the metals detected in the pond sediments, only the concentration of iron in pond#1 outflow exceeded available sediment-quality guidelines for protection of freshwater benthic organisms reported by Persaud and others (1993) for iron (Fe_2O_3 = 5.72 wt%) and manganese (MnO_2 = 0.142 wt%) and by MacDonald and others (2000) for arsenic (As = 33 ppm), chromium (Cr = 111 ppm), copper (Cu = 149 ppm), nickel (Ni = 48.6 ppm), lead (Pb = 128 ppm), and zinc (Zn = 459 ppm).

Considerations for Remediation of the Abandoned Mine Drainage at the ALPO-SBTU Test Site

Data on the characteristics of AMD sources at the ALPO-SBTU test site indicate the extent and severity of the AMD, as described above, and can be evaluated to identify possible passive treatment alternatives for the AMD sources (fig. 2). Combined with empirical tests on the rates of neutralization of AMD samples by locally available substrates, described below, site-specific strategies for remediation of

the AMD can be considered. In the case of the ALPO-SBTU test site, limestone and steel slag substrates were evaluated to determine their potential for use in AMD treatment.

Mineralogy and Chemistry of Limestone and Steel Slag for Cubitainer Tests

The mineralogical and chemical compositions of the limestone and steel slag used in cubitainer tests are summarized in tables 4 and 5. The limestone was dark gray with minor white and black veins. On the basis of XRD and chemical analyses of finely crushed samples (table 4), the limestone was composed of calcite ($CaCO_3$) with minor dolomite ($CaMg(CO_3)_2$), quartz (SiO_2), and muscovite ($KAl_3Si_3O_{10}(OH)_2$). The chemical analysis indicated the limestone contained mostly calcium (CaO = 42.58 wt%) with smaller amounts of magnesium (MgO = 6.38 wt%), silicon (SiO_2 = 7.15 wt%), aluminum (Al_2O_3 = 1.63 wt%), iron (Fe_2O_3 = 0.68 wt%), and potassium (K_2O = 0.56 wt%). The concentrations of calcium and magnesium equate to 76.0 wt% as $CaCO_3$ and 13.4 wt% as $MgCO_3$. However, the XRD patterns did not reveal dolomite peaks (Jane Hammarstrom, U.S. Geological Survey, written commun., 2004). Hence, dolomite and (or) magnesian calcite probably were present but not in sufficient quantity to be detected by XRD. Silicon, aluminum, and potassium in the limestone could be explained by the presence of quartz and small amounts of muscovite. Except for strontium (Sr = 905 ppm) and barium (Ba = 133 ppm), the limestone was relatively free of trace-element impurities (table 5). Trace amounts of strontium and barium could substitute for calcium in major carbonate minerals (Hanshaw and Back, 1979).

The steel slag had the appearance of porous cement or volcanic tuff; it was pale gray with a powdery white surface and numerous air vesicles ranging to 1 millimeter (mm) diameter. XRD analysis indicated the slag was composed predominantly of gehlenite ($Ca_2Al(AlSi)O_7$) and larnite (Ca_2SiO_4) with minor ettringite ($Ca_6Al_2(SO_4)_3(OH)_{12}\cdot 26\ H_2O$), gypsum ($CaSO_4\cdot 2H_2O$), and calcite (table 4). Non-crystalline phases (glasses) are probable in the slag because the peak-to-noise ratio on the XRD pattern was relatively low, suggesting an XRD-amorphous component (Jane Hammarstrom, U.S. Geological Survey, written commun., 2004). Additionally, zeolites could be present at low concentrations resulting from weathering of the slag minerals and glass (Nadine Piatak, U.S. Geological Survey, written commun., 2004). The calcium-silicate minerals and glasses of similar composition are typical of steel slag (Bayless and Schulz, 2003; Ceramics Research Center, 2003) and account for the bulk composition ($Ca_2SiO_4 = 2\ CaO + SiO_2$; $Ca_2Al(AlSi)O_7 = 2\ CaO + Al_2O_3 + SiO_2$). The white surface coating on the slag samples consisted of ettringite, gypsum, and calcite, which are common minerals in portland cement and alkaline fly ash (Myneni *et al*., 1998; Loop *et al*., 2003).

Like the limestone, the steel-slag sample contained mostly calcium (CaO = 42.69 wt%) with smaller amounts of magnesium (MgO = 3.57 wt%), iron (Fe_2O_3 = 0.47 wt%), and potassium (K_2O = 0.32 wt%),

but it also contained substantial silicon (SiO_2 = 32.79 wt%), aluminum (Al_2O_3 = 13.12 wt%), manganese (MnO = 0.68 wt%), and titanium (TiO_2 = 0.44 wt%) (table 4). Minor elements, such as iron, manganese, magnesium, and titanium (table 4) and trace elements, such as barium, strontium, and zirconium (table 5) could be in glass and (or) spinels, olivines, or pyroxenes that are present in concentrations too low to be confidently identified by XRD (Jane Hammarstrom, U.S. Geological Survey, written commun., 2004). Except for strontium (Sr = 662 ppm) and rubidium (Rb = 3 ppm), the slag contained higher concentrations of trace metals than the limestone, but it had equivalent or lower concentrations of most trace metals detected in the pond sediments (table 5).

The calcite, gypsum, and ettringite that were identified in the white surface coatings on the slag represent weathering products that could form by reaction of the calcium-silicate minerals or glass with acidic rainfall. For example, with prolonged exposure to atmospheric CO_2 (as H_2CO_3) and SO_3 (as H_2SO_4), larnite and gehlenite could spontaneously convert to calcite, gypsum, and ettringite:

$$Ca_2SiO_4\ (s) + 2\ H_2CO_3 \leftrightarrow 2\ CaCO_3\ (s) + SiO_2\ (am) + 2\ H_2O \tag{3}$$

$$Ca_2SiO_4\ (s) + 2\ H_2SO_4 + 2\ H_2O \leftrightarrow 2\ CaSO_4{\cdot}2H_2O\ (s) + SiO_2\ (am) \tag{4}$$

$$2\ Ca_2SiO_4\ (s) + Ca_2Al(AlSi)O_7\ (s) + 3\ H_2SO_4 + 29\ H_2O \leftrightarrow Ca_6Al_2(SO_4)_3(OH)_{12}{\cdot}26\ H_2O\ (s) + 3\ SiO_2\ (am). \tag{5}$$

In reactions (3) through (5), atmospheric acids represented by H_2CO_3 and H_2SO_4 are neutralized, and only solid products are formed. Crystalline and amorphous solids are denoted by (s) and (am), respectively.

The principal calcium-bearing minerals in limestone and steel slag also can be effective for neutralization of AMD as demonstrated by cubitainer tests described below. For example, the dissolution of calcite can increase pH, alkalinity (predominantly $HCO_3^- + OH^-$), and calcium concentration by the following reactions or some combination thereof:

$$CaCO_3\ (s) + 2\ H^+ \leftrightarrow Ca^{2+} + H_2O + CO_2 \tag{6}$$

$$CaCO_3\ (s) + H_2O + CO_2 \leftrightarrow Ca^{2+} + 2\ HCO_3^- \tag{7}$$

$$CaCO_3\ (s) + H^+ \leftrightarrow Ca^{2+} + HCO_3^-. \tag{8}$$

Hence, the stoichiometric dissolution of 1 mole $CaCO_3$ will produce 1 mole Ca^{2+} and up to 2 moles alkalinity as HCO_3^-. However, the relative quantities of H^+ reacted and alkalinity produced per mole Ca^{2+} released vary depending on the predominant reaction. According to Plummer and others (1979), reaction (6) is predominant at pH values less than 5, whereas reaction (7) is predominant at pH values greater than

6 and becomes increasingly important with increased Pco_2. At pH values of 5 to 6, both reactions (6) and (7) are important. Reaction (8) is an overall reaction derived by the addition of reactions (6) and (7).

As indicated by reactions (3) and (4), calcium-silicate minerals and associated glass in steel slag can neutralize acid, with calcite or gypsum as possible products. Nevertheless, with the addition of AMD to steel slag, solid calcium-bearing products initially would be undersaturated in the aqueous phase (leachate) and only "non-carbonate" alkalinity as OH^- would be produced. Initially, when low-pH AMD (pH 3 to 4) contacts larnite and gehlenite, only the component CaO is presumed to be reactive:

$$Ca_2SiO_4\ (s) + 2\ H^+ \leftrightarrow 2\ Ca^{2+} + 2\ OH^- + SiO_2\ (s) \tag{9}$$

$$Ca_2Al(AlSi)O_7\ (s) + 2\ H^+ \leftrightarrow 2\ Ca^{2+} + 2\ OH^- + SiO_2\ (s) + Al_2O_3\ (s) \tag{10}$$

In accordance with reactions (9) and (10), solid silica and alumina (or related phases) will accumulate as the pH and alkalinity increase. Without carbonate buffering, described below, the pH of slag leachate can increase to highly alkaline levels (pH > 9) (Ziemkiewicz and Skousen, 1998; Loop *et al*., 2003). Although dissolved silicon and aluminum concentrations could be limited by equilibrium with solid SiO_2, Al_2O_3, $Al(OH)_3$, and (or) kaolinite, the solubility of these phases increases at high pH (Ball and Nordstrom, 1991; Langmuir, 1997). At alkaline pH, the residual silica and alumina can dissolve:

$$SiO_2\ (s) + H_2O + OH^- \leftrightarrow H_3SiO_4^- \tag{11}$$

$$Al_2O_3\ (s) + 3\ H_2O + 2\ OH^- \leftrightarrow 2\ Al(OH)_4^-. \tag{12}$$

Hence, dissolved silica concentrations could be expected to increase at alkaline pH by a combination of reactions (9), (10), and (11). Nevertheless, concentrations of dissolved aluminum in solutions containing sulfate and calcium may actually decline with continued increases in pH owing to the precipitation of "alkaline" solids, such as ettringite:

$$6\ Ca^{2+} + 2\ Al(OH)_4^- + 3\ SO_4^{2-} + 4\ OH^- + 26\ H_2O \leftrightarrow Ca_6Al_2(SO_4)_3(OH)_{12}\cdot 26\ H_2O. \tag{13}$$

The precipitation of ettringite by reaction (13) represents a sink for alkalinity, calcium, aluminum, and sulfate concentrations in leachate. Likewise, the precipitation of other silicate, hydroxide, or carbonate solids, such as kaolinite, $Al(OH)_3$, $Fe(OH)_3$, and (or) $CaCO_3$, could remove alkalinity plus aluminum, iron, and (or) calcium in the leachate from steel slag or limestone.

The pH of a solution in contact with steel slag or limestone will be determined by the relative rates of mineral dissolution and precipitation and atmospheric exchange. Eventually, as atmospheric CO_2 diffuses into an alkaline solution, "bicarbonate alkalinity" will form:

$$CO_2 + OH^- \leftrightarrow HCO_3^-. \tag{14}$$

As reaction (14) proceeds to the right, pH will decline, but total alkalinity will be unaffected. Forward reactions (11) and (12) also could cause pH to decline without affecting the total alkalinity. Total alkalinity will be conserved if an equal number of moles of HCO_3^-, $H_3SiO_4^-$, or $Al(OH)_4^-$ is produced for each mole of OH^- reacted because these dissolved species have equal molar capacities to neutralize H^+. Hence, pH and alkalinity can increase by forward reactions (6) through (10), pH can decrease by forward reactions (11) through (13), and the precipitation of solids, such as forward reactions (1), (2), and (13) and the reverse of reaction (8), can cause pH and alkalinity to decrease.

As the steel slag ages in piles on the land surface or the leachate from steel slag equilibrates with the atmosphere, carbonate reactions become important. Eventually, equilibrium can be established with atmospheric CO_2 and various calcium-bearing minerals, including calcite, gypsum, and ettringite. Once established, these equilibrium reactions will buffer the pH and impose upper limits on the alkalinity and concentrations of dissolved calcium, sulfate, and aluminum. With the establishment of carbonate equilibrium, the neutralization capacity of the slag and (or) final composition of slag leachate (treated effluent) could approach that of limestone. Nevertheless, rates of neutralization could be distinctive for limestone and steel slag because of differences in their compositions, crystallinities, and surface areas.

Cubitainer Tests of Short-Term Reaction Rates

Cubitainer tests were conducted using AMD collected April 27 from sites 1 and Fe as the influent because these represented "end-member" compositions for the AMD at the ALPO-SBTU test site (table 2). Site 1 AMD was saturated with oxygen (DO = 11.2 mg/L) and contained moderate concentrations of dissolved aluminum (Al = 4.96 mg/L) and relatively low concentrations of dissolved iron and manganese (Fe = 0.96 mg/L, Mn = 1.48 mg/L). In contrast, site Fe AMD contained negligible dissolved oxygen (DO = 0.7 mg/L) but high concentrations of dissolved aluminum, iron, and manganese (Al = 9.31 mg/L, Fe = 71.3 mg/L, Mn = 4.40 mg/L). Aqueous speciation computations with the computer program WATEQ4F (Ball and Nordstrom, 1991) using measured pH, Eh, and analytical concentrations of chemical constituents (tables 2 and 3) indicated the dissolved iron proportion in the influent AMD from sites 1 and Fe on April 27, 2004, was 19 and 100 percent Fe^{II}, respectively.

The chemical evolution of the AMD from sites 1 and Fe with elapsed time in the cubitainers and the corresponding rates of reaction between the AMD and limestone or steel slag are summarized in tables 6, 7, 8, and 9 and figures 5, 6, 7, and 8. Generally, for all four test conditions (site 1 AMD + limestone, site 1 AMD + steel slag, site Fe AMD + limestone, site Fe AMD + steel slag), the pH, alkalinity, and calcium concentrations in the effluent increased (fig. 5, tables 6 and 7), whereas the acidity and concentrations of dissolved aluminum and iron progressively decreased (tables 7 and 8). After prolonged contact with steel

slag, manganese concentrations in the AMD from sites 1 and Fe declined to less than 1 mg/L; however, they did not change after equivalent contact with limestone. Nevertheless, manganese is not considered toxic at concentrations observed at the ALPO-SBTU test site (fig. 3G), and the regulation of manganese in coal-mine effluent is subject to debate (Kleinmann and Watzlaf, 1986). The concentration of magnesium increased for tests with limestone, while the concentration of silica increased for tests with steel slag (table 7). Except for increases in strontium and barium, which substitute for calcium in limestone and steel-slag minerals, and, to a lesser extent, chromium, titanium, lithium, and vanadium, the concentrations of most dissolved trace elements in effluent declined after prolonged contact with the limestone or slag (table 8). Hence, use of steel slag or limestone could be effective for increasing pH and alkalinity while decreasing concentrations of dissolved metals of concern.

The pH and dissolved calcium concentrations increased asymptotically with elapsed contact time (detention time) between the AMD and limestone or steel slag (figs. 5 and 6, table 6). These trends indicated progressive declines in the rates of dissolution of the limestone or steel slag with increased pH and an approach to “steady-state” or equilibrium conditions after approximately 2 weeks (336 hr). At each time step for the same AMD influent (site 1 or site Fe), the pH and concentrations of calcium were greater for tests with steel slag than limestone (fig. 5). At the start of the tests, the AMD influent from sites 1 and Fe had pH of 3.7 and 4.0, respectively. However, after only 1 hour of contact between the AMD and the limestone or slag substrates, the pH for all tests increased to values ranging from 4.9 to 5.1 (tables 6 and 7). After 8 hr of contact, three of the four tests produced net-alkaline effluent (net acidity < 0 for site 1 AMD with limestone and with steel slag and site Fe AMD with steel slag; net acidity > 0 for site Fe with limestone); all four effluents had pH values of 6.1 to 6.7. After 2 weeks of contact, the effluent for all tests was net alkaline, with pH values ranging from 6.5 to 7.4 for limestone and from 8.5 to 8.6 for steel-slag substrates.

The trends for alkalinity, net acidity, and dissolved iron were more complex than those for pH and calcium. Short-term trends for alkalinity production and acidity removal were distinctly different between the limestone and steel-slag tests with site Fe AMD and, to a lesser extent, with site 1 AMD (fig. 5, table 7). Generally, the pH and alkalinity increased progressively with elapsed contact time in cubitainers filled with limestone. Although the effluent contacting steel slag ultimately had the highest pH values and calcium concentrations, the highest alkalinities were produced by limestone (fig. 5, table 6). After 2 weeks contact with limestone, concentrations of calcium increased by 124 and 248 mg/L as $CaCO_3$ and alkalinity increased to 45.5 and 108 mg/L as $CaCO_3$ for tests with AMD from sites 1 and Fe, respectively (fig. 6, table 6). However, after 2 weeks contact with steel slag, concentrations of calcium increased by 466 and 730 mg/L as $CaCO_3$ and alkalinity peaked and then declined to 18.6 and 25.8 mg/L as $CaCO_3$ for tests

with AMD from sites 1 and Fe, respectively (fig. 6, table 6). If dissolved calcium resulted from dissolution of gypsum instead of calcite, gehlenite, or larnite, alkalinity would not increase with calcium concentration. Nevertheless, supersaturation with calcite could result because of increased calcium concentration at near-neutral or alkaline pH. The final effluents in contact with steel slag were supersaturated with calcite; those effluents in contact with limestone were undersaturated with calcite (table 9). Although apparent steady-state conditions had been achieved, the lack of equilibrium after 2 weeks of contact between the AMD and substrate implies that solution compositions could continue to change, albeit gradually, as carbonate reactions continued toward equilibrium conditions.

The initial rates of alkalinity production for site Fe AMD in contact with limestone or steel slag exceeded those for the same substrates in contact with site 1 AMD (fig. 5). Two possible explanations for this difference are (1) greater initial Pco_2 (table 9) and (2) greater initial "available" acidity as H^+ and Al^{3+} for site Fe AMD (59 mg/L as $CaCO_3$) compared to site 1 AMD (37 mg/L as $CaCO_3$). Elevated quantities of H^+ and CO_2 in the influent would tend to make the site Fe solution more aggressive or reactive than the site 1 solution. Other differences in the relative rates of change in pH, calcium, alkalinity, acidity, and metals concentrations for individual tests and among tests warrant additional discussion, but speculation on these differences is beyond the scope of this report.

Dissolved iron initially present as Fe^{III} precipitated rapidly after contact of the solutions with limestone or steel slag. Although the reaction with limestone had little effect on the concentration of any dissolved iron that was initially present as Fe^{II} for both AMD sources (table 7), dissolved Fe^{II} was substantially removed from the effluent in contact with steel slag. The steel-slag tests exhibited a rapid alkalinity increase, peaking at 24 hr for site Fe and at 72 hr for site 1, followed by a gradual decrease, presumably associated with the removal of the dissolved iron and manganese (fig. 5, tables 6 and 7). The decrease following the alkalinity peak indicates consumption of initial alkalinity and is consistent with an increased rate of oxidation of dissolved Fe^{2+} and Mn^{2+} at neutral or alkaline pH values. Higher rates of oxidation and hydrolysis are expected with increased pH and increased availability of oxygen (in pores and on surfaces) in the steel slag compared to the limestone substrate. Precipitate formed a pale orange coating on the steel slag and, eventually, the limestone substrates in the cubitainers.

The calcium and alkalinity concentration data for the first 7 hr of cubitainer testing and the maximum concentrations observed during the tests were used to compute first-order and second-order estimates of the concentration as a function of detention time (fig. 6, table 9). Generally, second-order models approximated the calcium concentration changes with detention time, whereas first-order models approximated the alkalinity concentration trends (fig. 6).

Cubitainer Estimates of Long-Term Performance

The results of computations to simulate long-term, decadal-scale mass decay and associated alkalinities for porous beds of limestone or steel-slag substrates for treatment of AMD from sites 1 and Fe (figs. 7 and 8) are based on the cubitainer concentration data and associated dissolution rate estimates (fig. 6, table 9). The initial masses of substrate for site 1 (8,800 ton = 8,000 tonne) and site Fe (500 ton = 455 tonne) were selected to achieve a minimum detention time of 8 hr and (or) a minimum net-alkalinity concentration of 20 mg/L for a 20-yr life span. Instead of the maximum flow rate measured for site Fe (0.9 gal/min), the flow rate for site 2 (18 gal/min) was used to estimate the size of the limestone or steel-slag bed for treatment of AMD from site Fe, because the site 2 flow measurement accounts for various additional diffuse seepage sources from the iron mound represented by site Fe influent. Decreases in the substrate mass and associated detention time with increased age of the treatment system were estimated on the basis of the concentration and flux of calcium as $CaCO_3$ at the maximum measured flow rate (table 2). As the limestone mass decreased with age, its total volume was assumed to decrease proportionally, whereas the porosity (42 volume percent (vol%)) and flow rates for site 1 (360 gal/min) and site Fe (18 gal/min) were assumed to remain constant. Calculations of detention time and corresponding concentrations and fluxes of alkalinity and calcium as $CaCO_3$ were repeated for sequential time steps to simulate the long-term performances.

The simulation of long-term performance of a treatment system on the basis of short-term cubitainer tests (figs. 7 and 8) should be considered only as a crude guide of possible field performance because few variables could be evaluated and (or) were assumed constant. Additional bench-scale testing using cubitainers or pilot-scale testing in the field could be helpful to refine models of long-term performance and to optimize the design of a treatment system. For example, tests could evaluate the effects of mixing or layering steel slag, limestone, and (or) compost on alkalinity production and metals removal. Such testing could evaluate the effects of particle sizes and mineral coatings on dissolution rates. Specific data on the potential for discharging solids from these packed beds and data on the porosity and permeability of different substrates also could be obtained.

Identification of Remediation Strategies at the ALPO-SBTU Test Site

Water-quality criteria for identification of AMD treatment alternatives (fig. 2) were considered with data on the flow and concentration of net acidity, dissolved oxygen, and dissolved iron, manganese, and aluminum to rank the AMD sources at the ALPO-SBTU test site by their pollutant loadings and to indicate possible passive-treatment alternatives. Sites 1, 7, and 6 had the largest loadings of dissolved metals among the AMD sources and were the first-, second-, and third-ranked pollutant sources, respectively, on

this basis (table 10). These sources had moderate flows (50 to 500 gal/min) with very high concentrations of dissolved aluminum (Al > 4 mg/L) and low-to-moderate concentrations of dissolved iron and manganese (1 < Fe ≤ 3 mg/L and 1 < Mn ≤ 2 mg/L). Because these and the other AMD sources at the ALPO-SBTU test site were net acidic, treatment steps or components that add alkalinity to the AMD and that facilitate removal of precipitated metal-rich solids from the alkalinity-producing system were identified as potentially appropriate (fig. 2, table 10). Because the AMD sources were oxygenated or contained more than 2 mg/L of dissolved aluminum, none met criteria for an ALD (Hedin *et al*., 1994). Cravotta and Trahan (1999) and Cravotta and Watzlaf (2002) suggested that aluminum concentrations less than 5 mg/L may be treated with underground anoxic or oxic limestone drains that are flushable. Hence, appropriate passive-treatment alternatives included the VFCW for most AMD sources or a flushable oxic limestone drain (OLD) and (or) open limestone channel (OLC) for AMD sources with low to moderate concentrations of metals (table 10) (Skousen *et al*., 1998; Cravotta and Trahan, 1999). In addition to the alkalinity-producing VFCW, OLD, or OLC, an aerobic pond or wetland was indicated as an additional component to facilitate iron and manganese oxidation and retention of precipitated solids (table 10). The combination of a VFCW and aerobic pond also was identified as a possible treatment option for treatment of combined AMD from sites 1, 1Fe, Fe, and 2 and from sites 3, 3B, 5, 6, and 7 based on their combined flows and volume-weighted average compositions (table 10).

Although beyond the scope of this investigation, the publicly available computer program AMDTreat (U.S. Office of Surface Mining Reclamation and Enforcement, 2002) could be used with the flow and water-quality data collected for this project to evaluate differences in costs among various possible passive-treatment and active-treatment scenarios. For example, a resource manager may wish to compare the costs for construction and maintenance of a passive-treatment system consisting of an OLC to collect the AMD, a VFCW as the primary source of alkalinity, and then an aerobic wetland to remove iron precipitate with an active treatment system that uses a hydraulic powered lime-dosing system to add alkalinity instead of a VFCW. Options to collect precipitated solids generated by passive or active AMD treatment also would need to be considered.

Although basic VFCW designs could be applicable for passive treatment of the AMD at the ALPO-SBTU test site, consideration could be given to innovative passive-system designs that direct flow upward through the treatment bed and that incorporate siphons for automatic flushing of solids to a pond (Vinci and Schmidt, 2001; Weaver *et al*., 2004; Schueck *et al*., 2004). Siphons and associated plumbing components could be incorporated in a modified VFCW design that includes or excludes the compost layer. Also, it may be advantageous to include a layer of steel slag on top of limestone for an upflow treatment system, on the basis of cubitainer testing that showed potential for lower alkalinity production

but higher pH of AMD treated with steel slag compared to limestone. The slag appears to offer greater short-term benefit with initially high pH, whereas the limestone appears to offer greater long-term benefit with greater alkalinity and possibly greater longevity. A combination of limestone and steel-slag substrates in the treatment system may take advantage of these differing, complementary properties. Burial of the limestone and steel-slag beds could be advantageous to increase Pco_2 and alkalinity concentrations and to minimize potential for entry of debris or animals.

Summary and Conclusions

This report, prepared by the U.S. Geological Survey (USGS) in cooperation with the National Park Service (NPS), describes field, laboratory, and computational methods that, when combined, can be used to assess possible remedial alternatives for abandoned mine drainage (AMD). During April-June, 2004, an assessment process was applied to AMD from bituminous coal deposits at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site (ALPO-SBTU) in Cambria County, Pennsylvania. The approach used at this test site could be applicable at other sites subject to drainage from abandoned coal or metal mines.

Data on the flow rates and composition of AMD, pond water and sediment, and the effluent and solids from cubitainers were used to assess the characteristics, possible ecological effects, and possible alternatives for remediation of the AMD at the ALPO-SBTU test site. During April 2004, the combined AMD sources had a total flow rate of 1,420 gallons per minute (gal/min) and flow-weighted averages for pH of 3.3, net acidity of 55 milligrams per liter (mg/L) as $CaCO_3$, and concentrations of dissolved sulfate and aluminum of 694 mg/L and 4.4 mg/L, respectively, which exceeded effluent criteria for active mines in Pennsylvania. Flow-weighted average concentrations of dissolved iron and manganese of 0.74 mg/L, and 1.2 mg/L, respectively, met these effluent criteria and met criteria for protection of freshwater aquatic organisms. The sum of iron, aluminum, and manganese loading for the combined AMD sources was 19.7 and 16.6 tons per year (ton/yr) on April 7 and 27, 2004, respectively, predominantly because of aluminum.

Elevated concentrations of dissolved trace metals were present in the AMD and downstream pond sediments at the ALPO-SBTU test site. Concentrations of cobalt, nickel, and zinc in the AMD samples consistently exceeded continuous concentration criteria for protection of freshwater aquatic life. Additionally, chromium, copper, and lead concentrations in several of the AMD samples exceeded relevant continuous concentration criteria. The pond sediments had as much as three times higher concentrations of aluminum, arsenic, and chromium, but equivalent or lower concentrations of iron, copper, nickel, lead, and zinc compared to median concentrations for streambed sediments across the conterminous United States.

Nevertheless, of the metals detected in the pond sediments, only the concentration of iron in pond#1 outflow exceeded available sediment-quality guidelines for protection of freshwater benthic organisms.

Limestone and steel slag were tested in the laboratory for their composition, approximate surface area, and potential to neutralize samples of the AMD from the ALPO-SBTU test site. The substrates had a similar particle-size distribution and identical calcium content (43 percent as calcium oxide); however, the limestone was composed of carbonates and the slag was composed of silicates. After a minimum of 8 hours contact between the AMD and limestone or steel slag in closed containers (cubitainers), net-neutral effluent was produced, and the concentrations of iron, aluminum, and most dissolved trace elements in effluent from the cubitainers declined relative to influent concentrations. The steel slag produced lower alkalinity concentrations but higher pH compared to limestone. Ultimately, the pH of AMD in contact with limestone or steel slag was maintained greater than 6.0 and less than 9.0. The cubitainer test results extrapolated over a decadal time scale indicated that passive treatment of the AMD at the ALPO-SBTU test site may be feasible with limestone and (or) steel slag substrates.

The process for evaluating AMD remedial strategies at the ALPO-SBTU test site involved the computation and ranking of the metal loadings during April 2004 for each of the AMD sources and a comparison of the data on AMD flow and chemistry (alkalinity, acidity, dissolved oxygen, ferric iron, aluminum) with published criteria for selection of passive-treatment technology. Although neutralization of the AMD by reaction with limestone was demonstrated with cubitainer tests, an anoxic limestone drain (ALD) was indicated as inappropriate for any AMD source at the test site because all had excessive concentrations of dissolved oxygen and (or) aluminum. One possible passive-treatment scenario that was identified for the individual or combined AMD sources involved an open limestone channel (OLC) to collect the AMD source(s), a vertical flow compost wetland (VFCW) to add alkalinity, and an aerobic wetland to facilitate iron and manganese oxidation and retention of precipitated solids. Using the AMDTreat computer program with the flow and water-quality data collected for this study, resource managers may compute and compare the costs for implementation, long-term operation, and maintenance of a passive-treatment system with those for an active-treatment system.

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Figure 1. Color aerial photograph of the Staple Bend Tunnel Unit of Allegheny Portage Railroad (ALPO-SBTU) National Historic Site, Pennsylvania, showing locations of water-quality sampling sites. Aerial photography mosaic created by North Carolina State University Center for Earth Observation. ALPO-SBTU Park boundary and trace of drainage are approximate.

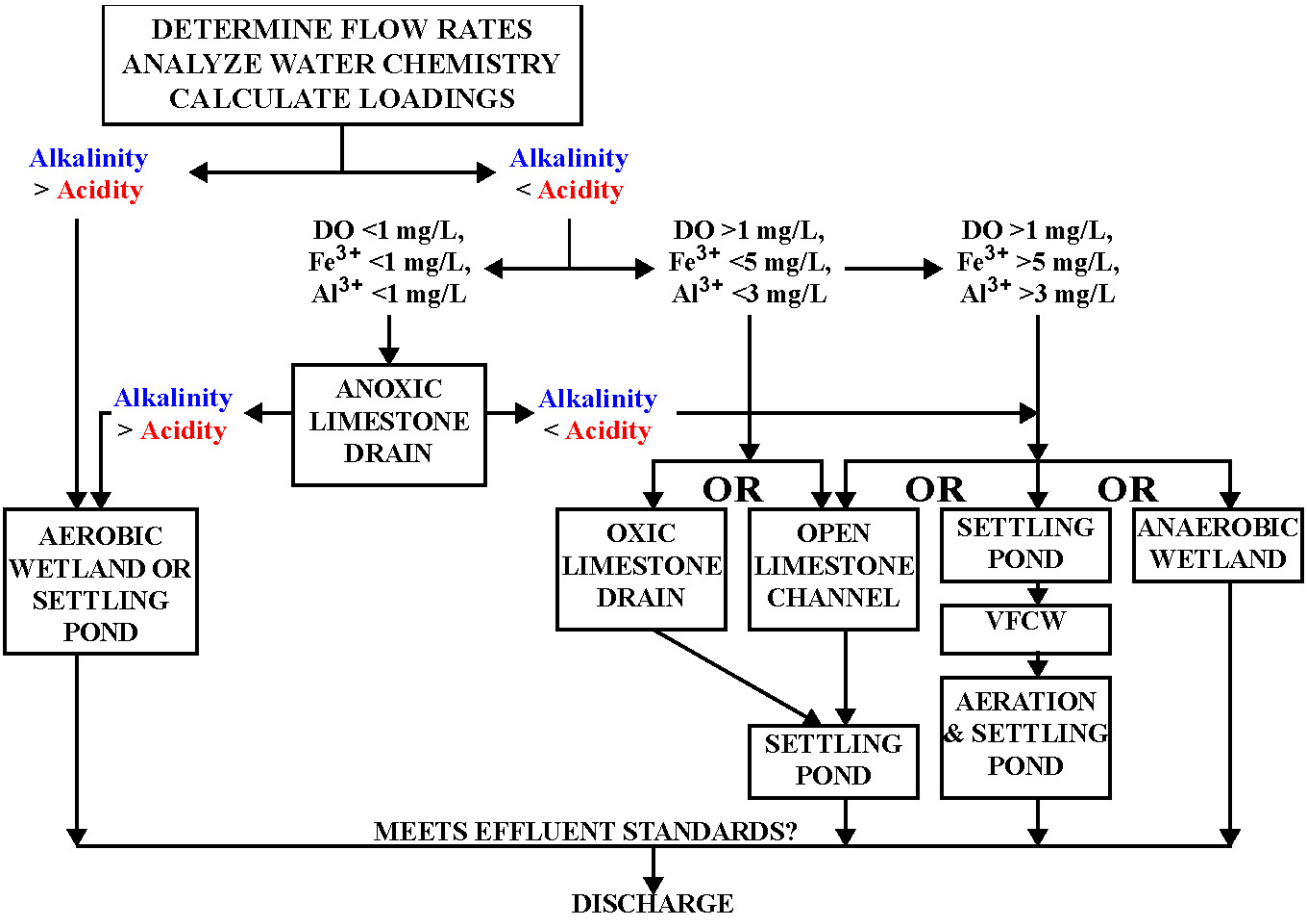

Figure 2. Flow chart for selection of passive treatment alternatives modified from Hedin and others (1994), Skousen and others (1998), and Pennsylvania Department of Environmental Protection (1999). Vertical flow compost wetland (VFCW) also known as SAPS or RAPS. Less than (<), greater than (>), milligrams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

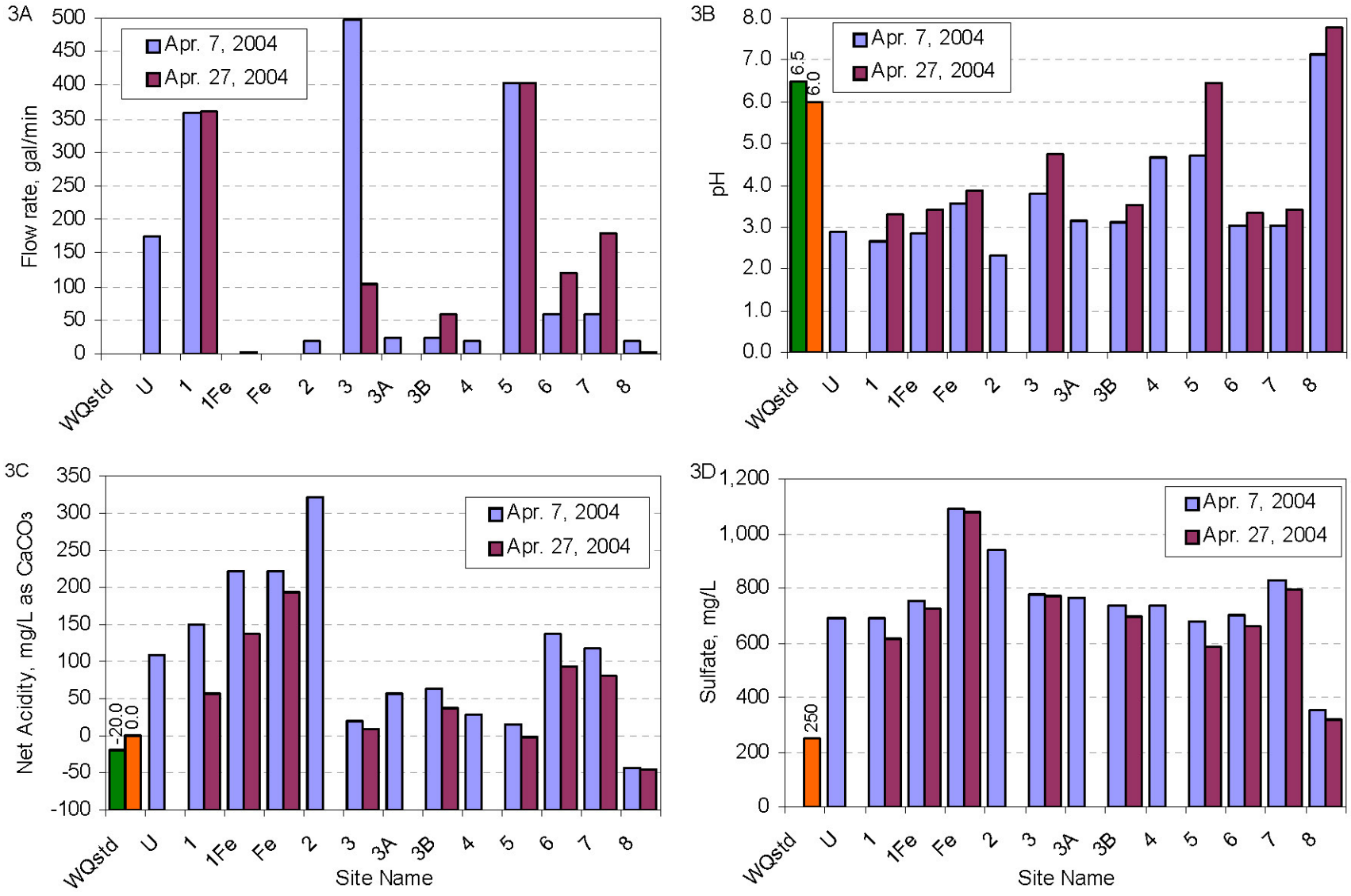

Figure 3. Flow rate and chemistry of abandoned mine drainage, April 7 and 27, 2004, Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania: A, Flow rate; B, pH; C, Net acidity (= acidity, computed - alkalinity); D, Sulfate concentration; E, Dissolved iron concentration; F, Dissolved aluminum concentration; G, Dissolved manganese concentration; H, Loading of dissolved iron, aluminum, and manganese. Water-quality standards (WQstd) at left; green is criteria continuous concentration for protection of freshwater aquatic life (U.S. Environmental Protection Agency, 2002a) and orange is standard for mine effluent (Commonwealth of Pennsylvania, 1998a).

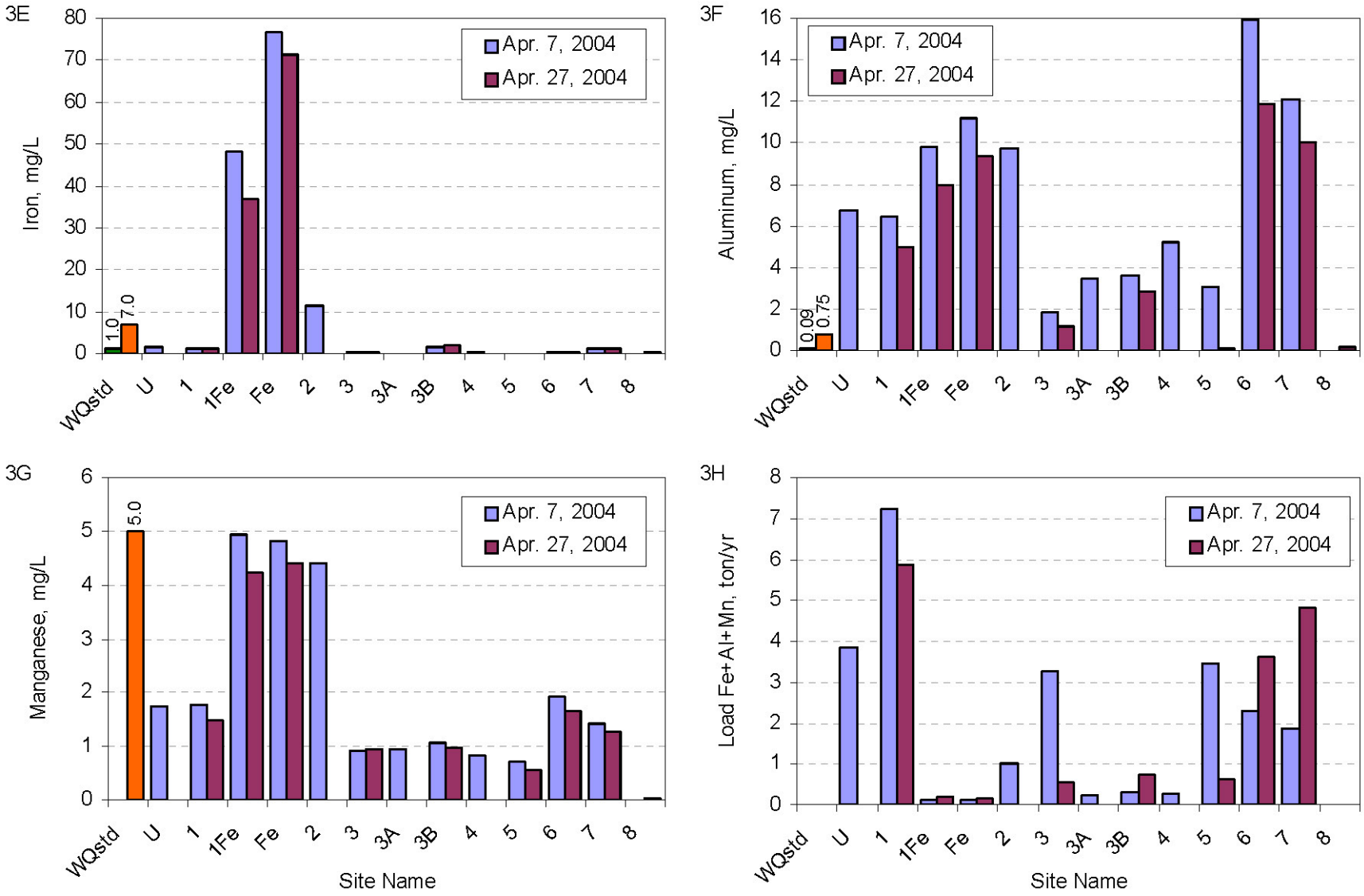

Figure 3. Flow rate and chemistry of abandoned mine drainage, April 7 and 27, 2004, Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania: A, Flow rate; B, pH; C, Net acidity (= acidity, computed - alkalinity); D, Sulfate concentration; E, Dissolved iron concentration; F, Dissolved aluminum concentration; G, Dissolved manganese concentration; H, Loading of dissolved iron, aluminum, and manganese. Water-quality standards (WQstd) at left; green is criteria continuous concentration for protection of freshwater aquatic life (U.S. Environmental Protection Agency, 2002a) and orange is standard for mine effluent (Commonwealth of Pennsylvania, 1998a) (continued).

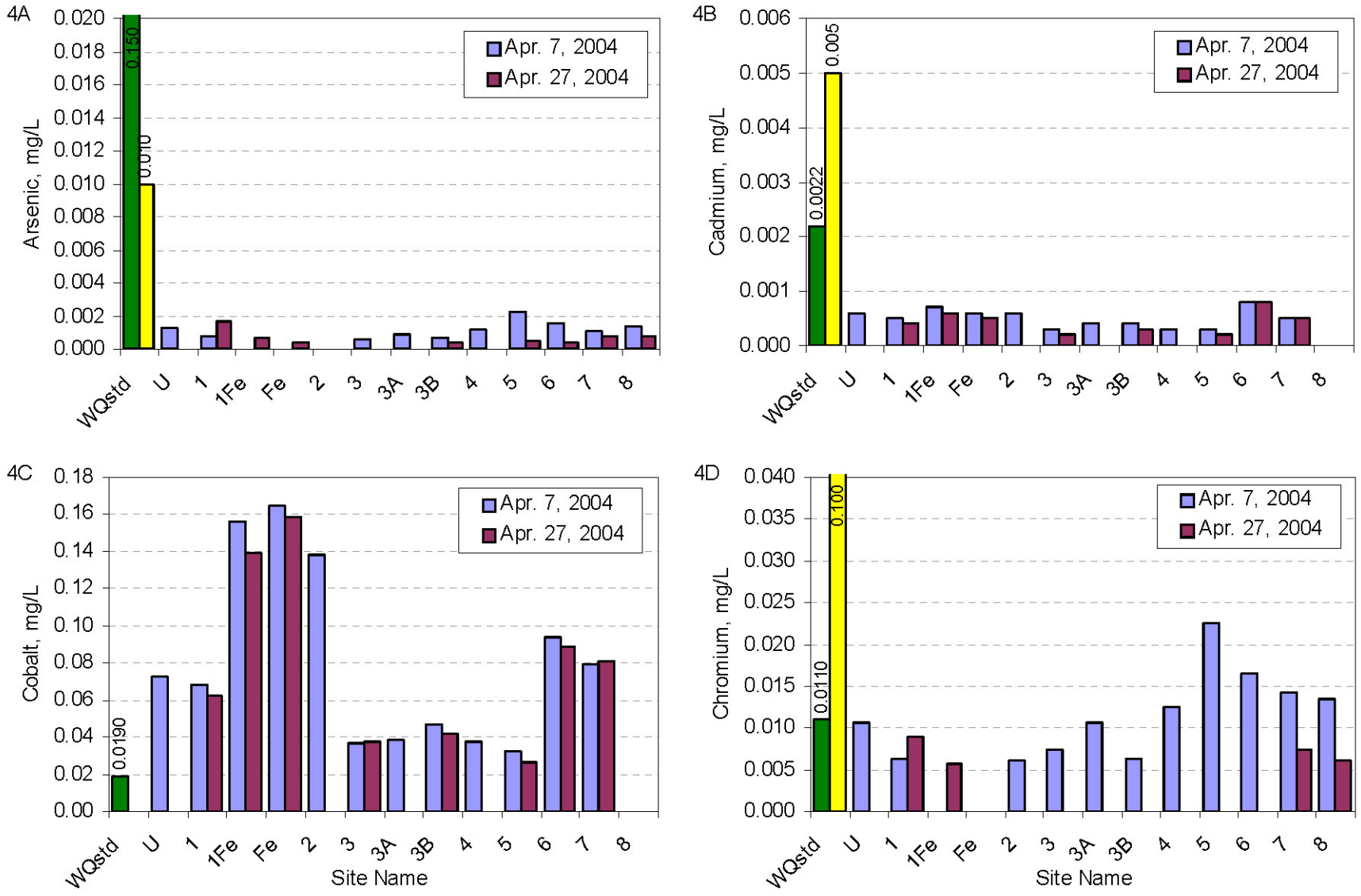

Figure 4. Trace-element concentrations in abandoned mine drainage, April 7 and 27, 200, Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania: A, Arsenic; B, Cadmium; C, Cobalt; D, Chromium; E, Copper; F, Lead; G, Nickel; H, Zinc. Water-quality standards (WQstd) at left; green is criteria continuous concentration for protection of freshwater aquatic life (U.S. Environmental Protection Agency, 2002a; Commonwealth of Pennsylvania, 2001b) and yellow is drinking-water standard (U.S. Environmental Protection Agency, 2002b).

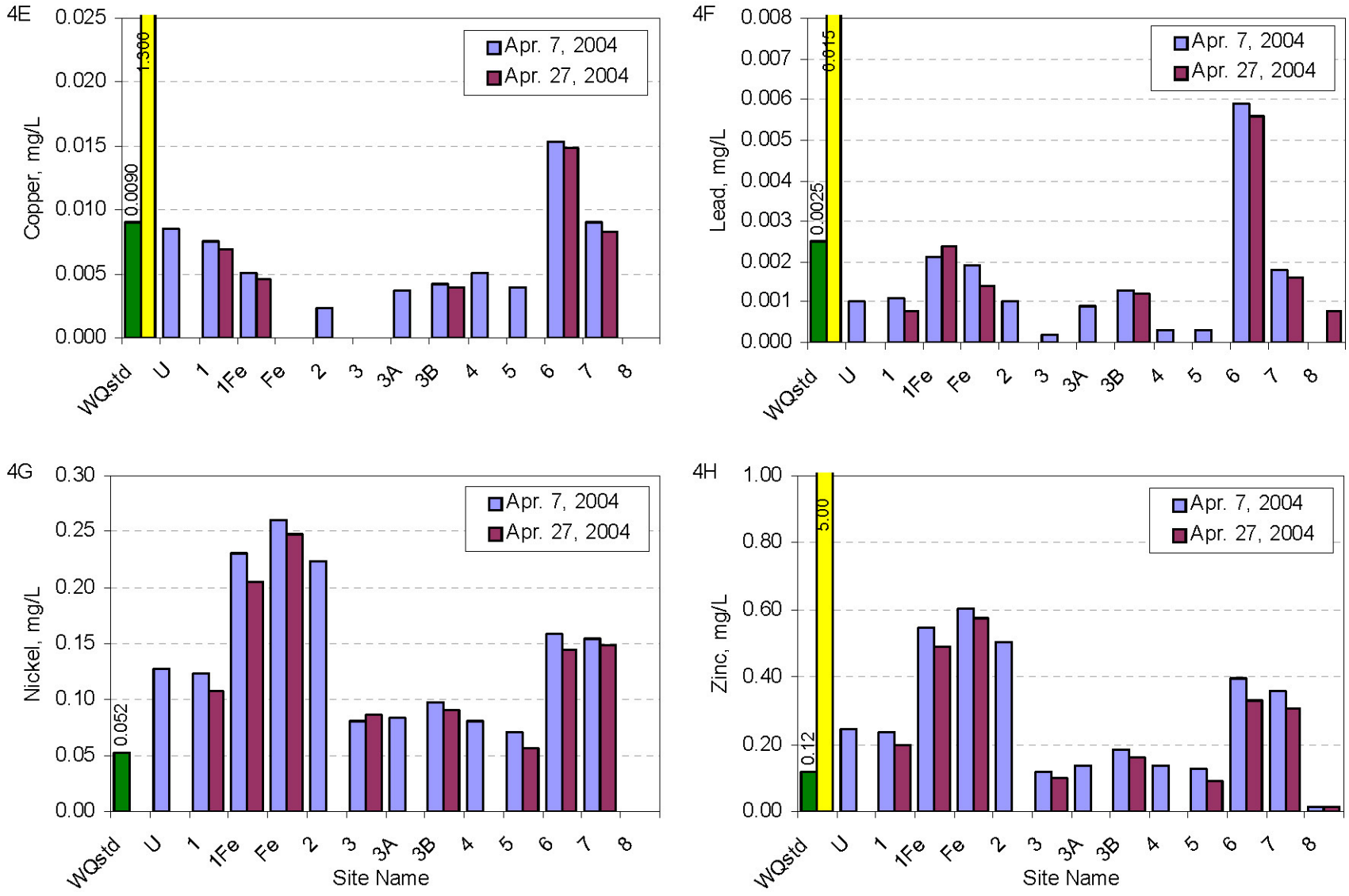

Figure 4. Trace-element concentrations in abandoned mine drainage, April 7 and 27, 2004, Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania: A, Arsenic; B, Cadmium; C, Cobalt; D, Chromium; E, Copper; F, Lead; G, Nickel; H, Zinc. Water-quality standards (WQstd) at left; green is criteria continuous concentration for protection of freshwater aquatic life (U.S. Environmental Protection Agency, 2002a; Commonwealth of Pennsylvania, 2001b) and yellow is drinking-water standard (U.S. Environmental Protection Agency, 2002b) (continued).

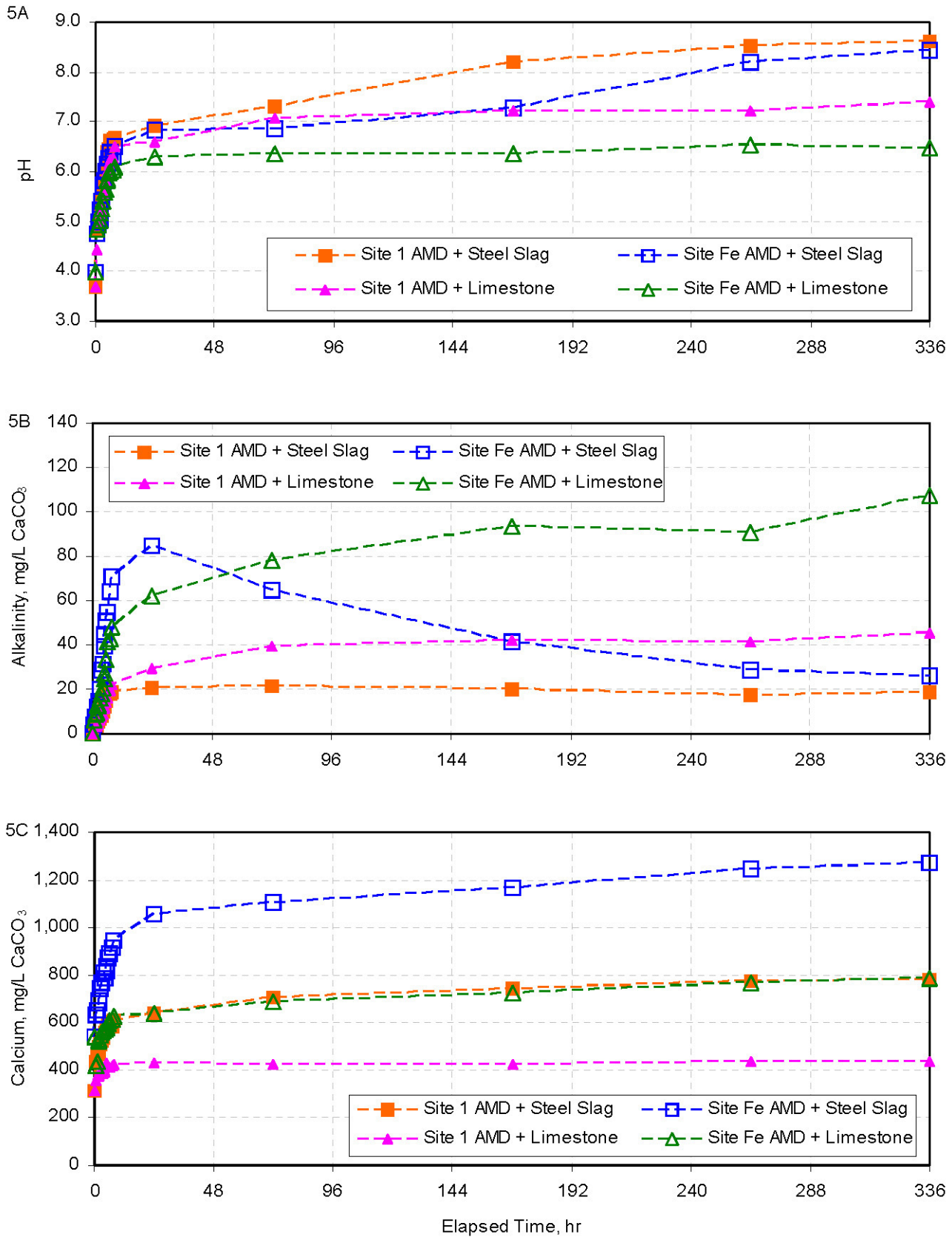

Figure 5. Rate of change in pH, alkalinity, and calcium concentrations of effluent from sites 1 and Fe at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania, during cubitainer tests: A, pH; B, Alkalinity concentration; C, Calcium concentration. Tests were conducted in May 2004 with 2 kilograms uncoated limestone or steel slag under closed, circulated conditions. Data on compositions of effluent from cubitainers are shown in tables 6-9. Data on composition of slag and limestone are shown in tables 4 and 5.

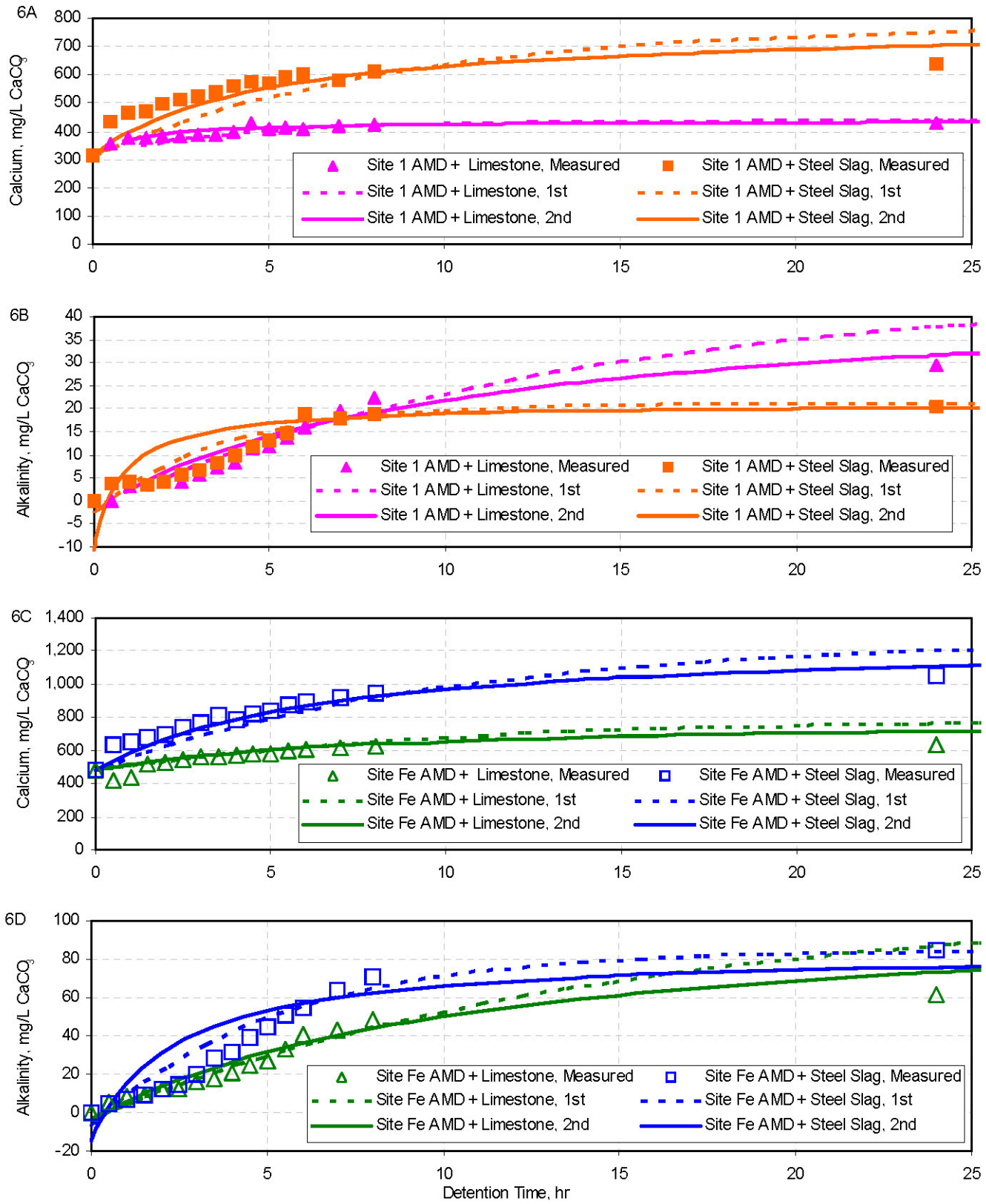

Figure 6. Measured and simulated effect of detention time on limestone or steel-slag treatment of abandoned mine drainage from sites 1 and Fe at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania, on the basis of cubitainer tests: A, Calcium concentration for site 1; B, Alkalinity concentration for site 1; C, Calcium concentration for site Fe; D, Alkalinity concentration for site Fe. Cubitainer test data are shown in figure 5 and tables 6 and 9. First-order curve dashed; second-order curve solid.

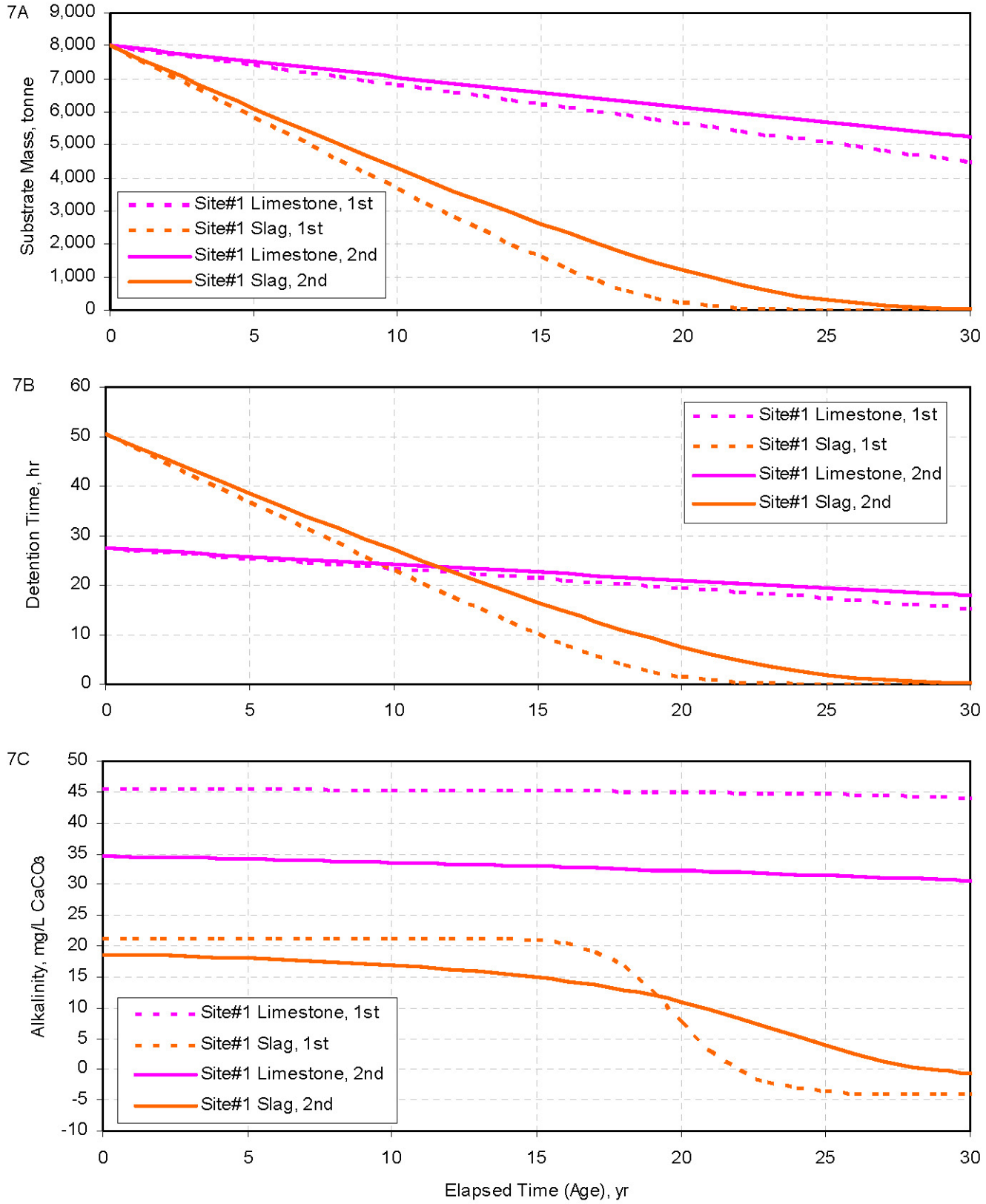

Figure 7. Simulation of long-term performance of limestone- or steel-slag substrates for treatment of abandoned mine drainage from site 1 at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania, on the basis of cubitainer test data and corresponding first-order or second-order rate estimates for limestone or steel slag dissolution and alkalinity production (table 9). Computations assume constant flow rate of 359 gal/min, limestone porosity of 0.42 , steel slag porosity of 0.55., and surface area of 0.72 cm^2/g for limestone and steel slag.

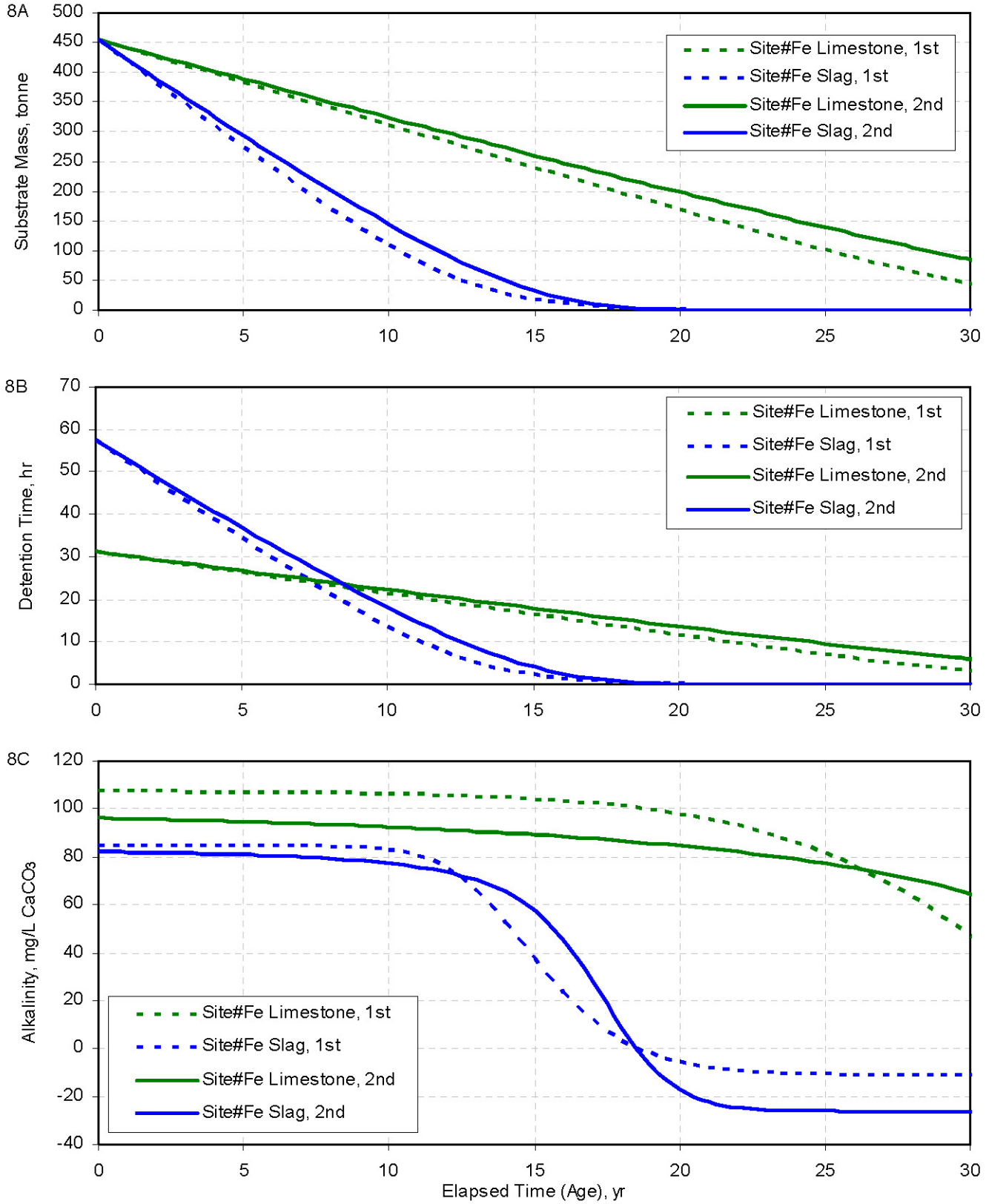

Figure 8. Simulation of long-term performance of limestone- or steel-slag substrates for treatment of abandoned mine drainage from site Fe at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania, on the basis of cubitainer test data and corresponding first-order or second-order rate estimates for limestone or steel slag dissolution and alkalinity production (table 9). Computations assume constant flow rate of 18 gal/min, limestone porosity of 0.42 , steel slag porosity of 0.55., and surface area of 0.72 cm^2/g for limestone and steel slag.

Table 1. Description of abandoned mine drainage and pond sites sampled in April 2004 for the assessment of the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania
[n.a., not applicable; latitude and longitude listed without degree, minute, and second symbols; 402228.7 represents 40°22'28.7" north latitude and 785105.8 represents 78°51'05.8" west longitude]

Local Identification Number[a]	Weir[b]	U.S. Geological Survey Station Identification[c]	Latitude	Longitude	Latitude, decimal degrees	Longitude, decimal degrees
Abandoned Mine Drainage						
U	1	402229078510601	402228.7	785105.8	40.3747	-78.8516
1	1A	402229078510602	402229.0	785110.4	40.3747	-78.8529
1Fe	n.a.	402232078510901	402231.7	785109.3	40.3755	-78.8526
Fe	n.a.	402223078511001	402223.4	785110.0	40.3732	-78.8528
2[d]	UPJ2	402225078511201	402225.1	785111.9	40.3737	-78.8533
3	3	402221078510701	402221.4	785107.4	40.3726	-78.8521
3A	n.a.	402221078510702	402220.2	785107.7	40.3723	-78.8522
3B	4	402221078510703	402219.1	785110.8	40.3720	-78.8530
4	n.a.	402214078510901	402213.9	785108.8	40.3705	-78.8525
5	5	402218078511101	402218.2	785111.2	40.3717	-78.8531
6	UPJ6	402212078511601	402212.0	785116.2	40.3700	-78.8545
7	7	402210078511701	402210.1	785117.2	40.3695	-78.8548
8	n.a.	402151078511001	402151.0	785110.0	40.3642	-78.8528
Upper Pond (Pond #1)						
Inflow (Pond1.In)	n.a.	402143078511204	402148.0	785115.6	40.3633	-78.8543
Al (Pond1.Al)	n.a.	402143078511201	402146.4	785116.0	40.3629	-78.8544
Middle (Pond1.Middle)	n.a.	402143078511202	402144.4	785117.0	40.3623	-78.8547
Outflow (Pond1.Out)	n.a.	402143078511203	402141.6	785119.6	40.3616	-78.8554
Lower Pond (Pond #2)						
Inflow (Pond2.In)	n.a.	402139078512401	402140.1	785120.3	40.3611	-78.8556
Outflow (Pond2.Out)	n.a.	402139078512501	402138.5	785125.4	40.3607	-78.8571

a. Local site identification numbers used to indicate site location in figure 1.
b. Weir identification from Kaktins and Carney (2002).
c. Formal USGS station identification number based on latitude and longitude.
d. Site 2 in this study was not identical to site 2 of Kaktins and Carney (2002). In this study, samples were collected at the outflow of a pipe that collected AMD from sites 1Fe, Fe, and other diffuse seepage from an iron mound before this seepage commingled with water from site 1. Although sampling points were only feet apart, Kaktins and Carney (2002) sampled the water at a culvert under the railroad, inclusive of sites 1, 1Fe, and Fe (sites 1 and 2 of this study).

Table 2. Flow rate, pH, net acidity, and concentrations of dissolved[a] constituents for abandoned mine drainage and pond samples collected April 7 and 27, 2004, for assessment of water quality at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania

[0, not flowing or dry; n.d., no data; <, less than; ≤, less than or equal to ; ≥, greater than or equal to; gal/min, gallons per minute; mg/L, milligrams per liter; ton/yr, tons per year]

Local Site Name	Flow rate (gal/min)		pH, Field (units)		Net Acidity[b] (mg/L $CaCO_3$)		Dissolved Oxygen (mg/L)		Dissolved Nitrate (mg/L N)		Dissolved Sulfate (mg/L SO_4)		Dissolved Silica (mg/L SiO_2)		Dissolved Aluminum (mg/L Al)		Dissolved Iron (mg/L Fe)		Dissolved Manganese (mg/L Mn)		Dissolved Al, Fe, and Mn load[c] (ton/yr)	
	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27
Abandoned Mine Drainage																						
U	175	n.d.	2.9	n.d.	109	n.d.	10.3	n.d.	0.35	n.d.	689	n.d.	22.2	n.d.	6.77	n.d.	1.43	n.d.	1.75	n.d.	3.83	n.d.
1	359	360	2.7	3.4	150	51	10.4	11.2	.37	.53	687	612	21.6	18.8	6.40	4.96	1.01	0.96	1.78	1.48	7.24	5.85
1Fe	.9	1.8	2.8	3.3	222	142	1.8	1.0	<.04	<.04	754	722	32.1	29.7	9.82	8.00	48.2	36.7	4.95	4.23	.11	.2
Fe	.4	.9	3.6	3.9	221	194	1.2	.7	<.05	<.05	1,090	1,080	34.7	31.0	11.2	9.31	76.4	71.3	4.82	4.40	.11	.15
2	18	n.d.	2.3	n.d.	322	n.d.	9.7	n.d.	.09	n.d.	940	n.d.	32.5	n.d.	9.76	n.d.	11.2	n.d.	4.40	n.d.	1.00	n.d.
3	498	104	3.8	4.7	20	8	10.2	10.5	.55	.56	779	773	14.3	12.7	1.81	1.16	.25	.21	.93	.93	3.27	.53
3A	23	n.d.	3.2	n.d.	57	n.d.	8.8	n.d.	.54	n.d.	763	n.d.	15.7	n.d.	3.47	n.d.	.19	n.d.	.95	n.d.	.23	n.d.
3B	24	60	3.1	3.5	63	36	9.6	9.9	.50	.54	738	697	17.1	15.5	3.62	2.84	1.60	1.75	1.06	.98	.33	.74
4	20	n.d.	4.7	n.d.	29	n.d.	10.5	n.d.	.48	n.d.	737	n.d.	15.2	n.d.	5.18	n.d.	.21	n.d.	.83	n.d.	.27	n.d.
5	404	404	4.7	6.5	16	-2	10.7	11.0	.52	.55	676	587	13.6	10.2	3.04	.06	.13	.11	.72	.55	3.45	.64
6	58	120	3.1	3.3	137	93	11.1	11.2	.71	.67	700	659	30.5	27.6	15.9	11.9	.21	.19	1.93	1.66	2.31	3.62
7	58	180	3.0	3.4	118	80	10.5	10.7	.66	.63	828	792	26.3	23.3	12.1	10.0	1.08	.96	1.42	1.28	1.87	4.83
8	20	3.1	7.1	7.8	-44	-45	11.9	11.0	.92	.75	351	317	5.8	5.9	<.02	.12	<.10	.23	<.01	.03	<.01	<.01
Upper Pond (Pond #1)																						
Inflow	260	324	3.2	3.6	90	60	11.4	11.1	0.74	0.68	536	499	22.7	24.0	9.25	7.62	0.45	0.60	1.48	1.46	6.39	6.88
Al	n.d.	n.d.	n.d.	3.3	n.d.	47	n.d.	9.9	n.d.	.77	n.d.	345	n.d.	14.9	n.d.	3.47	n.d.	.30	n.d.	.74	n.d.	n.d.
Middle	n.d.	n.d.	n.d.	3.5	n.d.	42	n.d.	9.7	n.d.	.72	n.d.	396	n.d.	17.3	n.d.	4.57	n.d.	.45	n.d.	1.00	n.d.	n.d.
Outflow	n.d.	n.d.	3.8	3.5	38	41	11.2	9.3	.96	.84	356	358	15.7	15.7	5.07	3.94	.25	.41	.84	.91	n.d.	n.d.
Lower Pond (Pond #2)																						
Inflow	n.d.	n.d.	3.7	4.0	33	30	11.1	7.2	1.01	0.80	334	360	15.1	16.1	4.12	4.17	<0.10	0.13	0.77	0.91	n.d.	n.d.
Outflow	n.d.	n.d.	n.d.	3.6	n.d.	41	n.d.	7	n.d.	.74	n.d.	383	n.d.	18.3	n.d.	4.54	n.d.	.19	n.d.	1.03	n.d.	n.d.
Criteria for Drinking Water, Mine Effluent, and Protection of Freshwater Aquatic Life[d]																						
Drink	n.d.		6.5≤pH≤8.5		n.d.		n.d.		<10.0		<250		n.d.		<0.20		<0.30		<0.05		n.d.	
Mine	n.d.		6.0≤pH≤9.0		<0		n.d.		n.d.		<250		n.d.		<0.75		<7.00		<5.0		n.d.	
CCCF	n.d.		6.5≤pH≤9.0		≤-20		≥5.0		n.d.		n.d.		n.d.		<0.087		<1.00		n.d.		n.d.	

a. Concentration data for dissolved oxygen are for in-situ measurement. Concentration and loading data for "dissolved" nitrate, sulfate, and metals are for filtered (0.45-micrometer pore size) samples.

b. Net acidity = computed acidity - measured alkalinity. Positive value indicates acidity exceeds alkalinity and the ultimate pH after complete oxidation and hydrolysis of metals will be acidic (pH < 4.5). Negative value indicates alkalinity exceeds acidity and the ultimate pH will be near neutral or alkaline (pH > 6). Acidity computed on the basis of the pH and dissolved metals concentrations where C_{Al}, C_{Fe}, and C_{Mn} indicate aluminum, iron, or manganese concentration, respectively, in milligrams per liter as: $Acidity_{computed}$ (mg/L $CaCO_3$) = $50 \cdot (10^{(3-pH)} + 3C_{Al}/26.98 + 2 \cdot C_{Fe}/55.85 + 2C_{Mn}/54.94)$.

c. Dissolved metals loading was computed as the product of the instantaneous flow rate (Q) and the sum of concentrations of the metals as: Metal load = $f \cdot Q \cdot (C_{Al} + C_{Fe} + C_{Mn})$, where C with a subscript Al, Fe, or Mn indicates aluminum, iron, or manganese concentration, respectively. For flow rate in gallons per minute and concentrations in milligrams per liter, the conversion factor, f = 5.45, yields daily loading in grams (g/d), whereas f = 0.00219, yields annual loading in short tons (ton/yr).

d. Drink = drinking-water standard (U.S. Environmental Protection Agency, 2002b). Mine = mine effluent standard (Commonwealth of Pennsylvania, 1998a). CCCF = criteria continuous concentration for protection of freshwater aquatic life based on hardness of 100 mg/L; values for arsenic, cadmium, chromium, copper, lead, nickel, and zinc from U.S. Environmental Protection Agency (2002a) and for cobalt from Commonwealth of Pennsylvania (2001b).

Table 3. Concentrations of dissolved[a] trace elements in abandoned mine drainage and pond samples collected April 7 and 27, 2004, for assessment of water quality at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania
[n.d., no data; <, less than; mg/L, milligrams per liter]

Local Site Name	Dissolved Arsenic (mg/L As)		Dissolved Cadmium (mg/L Cd)		Dissolved Cobalt (mg/L Co)		Dissolved Chromium (mg/L Cr)		Dissolved Copper (mg/L Cu)		Dissolved Lead (mg/L Pb)		Dissolved Nickel (mg/L Ni)		Dissolved Zinc (mg/L Zn)	
	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27	4/07	4/27
Abandoned Mine Drainage																
U	0.0013	n.d.	0.0006	n.d.	0.0721	n.d.	0.0107	n.d.	0.0086	n.d.	0.0010	n.d.	0.127	n.d.	0.244	n.d.
1	.0008	0.0017	.0005	0.0004	.0680	0.0620	.0062	0.0090	.0076	0.0069	.0011	0.0008	.123	0.107	.238	.200
1Fe	<.0003	.0007	.0007	.0006	.156	.139	<.0050	.0056	.0051	.0046	.0021	.0024	.231	.205	.549	.492
Fe	<.0003	.0004	.0006	.0005	.165	.159	<.0050	<.0050	<.0020	<.0020	.0019	.0014	.261	.247	.605	.574
2	<.0003	n.d.	.0006	n.d.	.138	n.d.	.0060	n.d.	.0023	n.d.	.0010	n.d.	.224	n.d.	.504	n.d.
3	.0006	<.0003	.0003	.0002	.0365	.0379	.0073	<.0050	<.0020	<.0020	.0002	<.0001	.081	.086	.116	.101
3A	.0009	n.d.	.0004	n.d.	.0388	n.d.	.0106	n.d.	.0037	n.d.	.0009	n.d.	.084	n.d.	.139	n.d.
3B	.0007	.0004	.0004	.0003	.0465	.0418	.0063	<.0050	.0042	.0039	.0013	.0012	.098	.090	.186	.161
4	.0012	n.d.	.0003	n.d.	.0378	n.d.	.0126	n.d.	.0051	n.d.	.0003	n.d.	.080	n.d.	.139	n.d.
5	.0022	.0005	.0003	.0002	.0327	.0264	.0225	<.0050	.0040	<.0020	.0003	<.0001	.071	.057	.126	.089
6	.0016	.0004	.0008	.0008	.0937	.0884	.0165	<.0050	.0154	.0148	.0059	.0056	.159	.145	.394	.330
7	.0011	.0008	.0005	.0005	.0796	.0808	.0143	.0073	.0090	.0083	.0018	.0016	.154	.149	.359	.308
8	.0014	.0008	<.0001	<.0001	<.0001	.0003	.0134	.0060	<.0020	<.0020	<.0001	.0008	<.003	<.003	.013	.016
Upper Pond (Pond #1)																
Inflow	0.0011	0.0019	0.0006	0.0008	0.0667	0.0679	0.0116	0.0173	0.0103	0.0108	0.0037	0.0034	0.113	0.111	0.268	0.253
Al	n.d.	.0015	n.d.	.0004	n.d.	.0312	n.d.	.0166	n.d.	.0047	n.d.	.0017	n.d.	.054	n.d.	.134
Middle	n.d.	.0044	n.d.	.0005	n.d.	.0421	n.d.	.0476	n.d.	.0063	n.d.	.0019	n.d.	.066	n.d.	.170
Outflow	.0011	.0017	.0005	.0005	.0359	.0378	.0110	.0190	.0054	.0054	.0026	.0018	.063	.064	.149	.148
Lower Pond (Pond #2)																
Inflow	0..0013	0.0018	0.0004	0.0005	0.0320	0.0377	0.0125	0.0195	0.0036	0.0046	0.0008	0.0014	0.053	0.064	0.147	0.151
Outflow	n.d.	.0034	n.d.	.0004	n.d.	.0443	n.d.	.0344	n.d.	.0050	n.d.	.0012	n.d.	.069	n.d.	.161
Criteria for Drinking Water, Mine Effluent, and Protection of Freshwater Aquatic Life[b]																
Drink	<0.0100		<0.0050		n.a.		<0.1000		<1.300		<0.0150		n.a.		<5.000	
Mine	n.a.		n.a.		n.a.		n.a.		n.a.		n.a.		n.a.		n.a.	
CCCF	<0.1500		<0.0220		<0.0190		<0.0110		<0.0090		<0.0025		<0.052		<0.120	

a. Concentration data are for “dissolved” constituents in filtered (0.45-micrometer pore size) samples.

b. Drink = drinking-water standard (U.S. Environmental Protection Agency, 2002b). Mine = mine effluent standard (Commonwealth of Pennsylvania, 1998a). CCCF = criteria continuous concentration for protection of freshwater aquatic life; values for arsenic, cadmium, chromium, copper, lead, nickel, and zinc from U.S. Environmental Protection Agency (2002a) and for cobalt from Commonwealth of Pennsylvania (2001b).

Table 4. Major mineralogical and chemical composition of pond sediment samples collected April 27, 2004, from the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania, and steel slag and limestone obtained from the vicinity[a] and used in cubitainer tests
[tr, trace; wt%, weight percent]

Sample Identification	Minerals[b]	Constituent Concentration as Oxide in Weight Percent[c,d]											
		SiO_2	Al_2O_3	Fe_2O_3	MnO	MgO	CaO	Na_2O	K_2O	TiO_2	P_2O_5	Loss on Ignition	TOTAL
Pond #1 Inflow	quartz, muscovite, kaolinite, chlorite, amorphous aluminum & iron hydroxides	56.30	18.97	5.65	0.07	0.92	0.22	0.26	2.74	1.06	0.11	12.62	98.92
Pond #1 Outflow	quartz, muscovite, kaolinite, chlorite, amorphous aluminum & iron hydroxides	52.79	18.66	6.17	0.02	0.95	0.25	0.20	2.77	0.98	0.13	15.68	98.61
Pond #2 Inflow	quartz, muscovite, kaolinite, chlorite, amorphous aluminum & iron hydroxides	49.19	13.45	3.19	0.01	0.48	0.18	0.18	1.58	0.87	0.12	29.59	98.84
Steel slag	gehlenite, larnite, ettringite(tr), gypsum(tr), calcite(tr)	32.79	13.12	0.47	0.68	3.57	42.69	0.13	0.32	0.44	0.02	4.52	98.75
Limestone	calcite, dolomite(tr), quartz(tr), muscovite(tr)	7.15	1.63	0.68	0.01	6.38	42.58	0.08	0.56	0.07	0.02	39.66	98.82

a. Steel slag was aged blast-furnace byproduct from Johnstown, Pa.; limestone was from Ashcom Quarry, near Everett, Bedford County, Pa.

b. Mineralogy identified by X-ray diffraction (XRD): Quartz (SiO_2), muscovite ($KAl_3Si_3O_{10}(OH)_2$), kaolinite ($Al_2Si_2O_5(OH)_4$), chlorite ($Mg_5Al_2Si_3O_{10}(OH)_8$), gypsum ($CaSO_4 \cdot 2H_2O$), gehlenite ($Ca_2Al(AlSi)O_7$), larnite (Ca_2SiO_4), ettringite ($Ca_6Al_2(SO_4)_3(OH)_{12} \cdot 26\ H_2O$), calcite ($CaCO_3$), dolomite ($CaMg(CO_3)_2$). Trace quantities of a mineral (less than 5 percent) or poorly crystalline or amorphous materials may not be detected by XRD owing to low peak to background ratios for such phases (J. M. Hammarstrom, U.S. Geological Survey, written commun., 2004).

c. Major elements analyzed by inductively coupled plasma atomic emission spectroscopy after lithium metaborate/tetraborate fusion. Loss on ignition indicates volatile components such as H_2O, CO_2, and SO_3.

d. Comparable values for median concentrations in streambed sediments across the conterminous United States were reported by Rice (1999) for aluminum (Al_2O_3 = 12.09 wt%) and iron (Fe_2O_3 = 5.00 wt%). Sediment-quality guidelines for protection of benthic freshwater organisms were reported by Persaud and others (1993) for iron (Fe_2O_3 = 5.72 wt%) and manganese (MnO = 0.142 wt%).

Table 5. Trace-element composition of pond sediment samples collected April 27, 2004, from the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania, and steel slag and limestone obtained from the vicinity[a] and used in cubitainer tests
[<, less than; ppm, parts per million]

Sample Identification	Element Concentration in Parts per Million[b,c]														
	Ag	As	Ba	Be	Bi	Ce	Co	Cr	Cs	Cu	Dy	Er	Eu	Ga	Gd
Pond #1 Inflow	<0.5	15	583	3	<0.4	93.3	13	75	7.0	23	5.8	3.7	1.52	22	6.3
Pond #1 Outflow	<.5	10	603	3	<.4	92.4	21	79	6.7	27	6.0	3.8	1.60	22	6.6
Pond #2 Inflow	<.5	21	433	3	<.4	63.9	28	110	4.4	50	6.1	3.6	1.37	14	6
Steel slag	<.5	<5	993	8	<.4	140	<1	22	<.5	17	8.4	5.6	2.44	1	8.7
Limestone	<.5	<5	133	1	<.4	8.7	<1	<20	.5	17	.6	.4	.17	2	.6
	Ge	Hf	Ho	In	La	Lu	Mo	Nb	Nd	Ni	Pb	Pr	Rb	Sb	Sc
Pond #1 Inflow	2	5.8	1.2	<0.2	44.5	0.50	<2	19	38.5	33	10	10.2	130	0.7	19
Pond #1 Outflow	1	5.2	1.2	<.2	44.6	.50	<2	18	39.3	44	10	10.5	128	<.5	19
Pond #2 Inflow	1	5.4	1.2	<.2	30.3	.50	<2	15	28.4	25	<5	7.22	78	1.0	20
Steel slag	<1	4.2	1.8	<.2	44.4	.67	<2	1	41.9	30	<5	10.3	3	<.5	19
Limestone	<1	.9	.1	<.2	4.1	.05	<2	1	3.7	<20	<5	.95	16	<.5	2
	Sm	Sn	Sr	Ta	Tb	Th	Tl	Tm	U	V	W	Y	Yb	Zn	Zr
Pond #1 Inflow	7.9	5	94	1.5	1.1	13.7	1.3	0.54	3.9	119	2	31	3.4	52	184
Pond #1 Outflow	8.0	5	90	1.4	1.1	13.4	1.5	.56	3.7	118	2	31	3.4	110	171
Pond #2 Inflow	6.6	9	75	1.2	1.1	11.5	1.6	.55	4.8	89	2	28	3.4	198	179
Steel slag	9.2	2	662	.2	1.5	10.3	<.1	.81	8.5	8	<1	53	4.8	<30	149
Limestone	.8	2	905	<.1	.1	1.2	<.1	.05	.8	<5	<1	4	.3	<30	34

a. Steel slag was aged blast-furnace byproduct from Johnstown, Pa.; limestone was from Ashcom Quarry, near Everett, Bedford County, Pa.

b. Trace elements analyzed by inductively coupled plasma emission mass spectrometry after lithium metaborate/tetraborate fusion.

c. Comparable values for median concentrations in streambed sediments across the conterminous United States were reported by Rice (1999) for arsenic (As = 6.3 ppm), chromium (Cr = 64 ppm), copper (Cu = 27 ppm), nickel (Ni = 27 ppm), lead (Pb = 27 ppm), and zinc (Zn = 110 ppm). Sediment-quality guidelines for protection of benthic freshwater organisms were reported by MacDonald and others (2000) for arsenic (As < 33 ppm), chromium (Cr < 111 ppm), copper (Cu < 149 ppm), nickel (Ni < 48.6 ppm), lead (Pb < 128 ppm), and zinc (Zn < 459 ppm).

Table 6. Cubitainer[a] test data for pH, alkalinity, and calcium concentrations[b] in effluent from reaction between steel slag or limestone with abandoned mine drainage (AMD) from Sites 1 and Fe at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania

[hr, hour; mg/L, milligrams per liter; additional data for shaded cells, including calcium analysis by inductively coupled plasma emission mass spectrometry, are shown in subsequent tables]

Elapsed Time (hr)	Site 1 AMD + Slag			Site 1 AMD + Limestone			Site Fe AMD + Slag			Site Fe AMD + Limestone		
	pH, Lab (units)	Alkalinity (mg/L as $CaCO_3$)	Calcium[c] (mg/L as $CaCO_3$)	pH, Lab (units)	Alkalinity (mg/L as $CaCO_3$)	Calcium (mg/L as $CaCO_3$)	pH, Lab (units)	Alkalinity (mg/L as $CaCO_3$)	Calcium (mg/L as $CaCO_3$)	pH, Lab (units)	Alkalinity (mg/L as $CaCO_3$)	Calcium (mg/L as $CaCO_3$)
0.0	3.7	0.0	312	3.7	0.0	312	4.0	0.0	540	4.0	0.0	540
0.5	4.8	3.8	434	4.4	0.0	358	4.8	4.3	630	4.8	5.7	420
1.0	5.1	4.2	464	5.0	3.1	374	5.0	7.2	652	4.9	8.8	434
1.5	5.2	3.6	472	5.0	3.6	374	5.1	9.4	680	5.0	9.4	520
2.0	5.4	4.1	498	5.2	4.7	380	5.2	12.0	694	5.2	12.6	526
2.5	5.6	5.6	512	5.2	4.2	384	5.4	14.4	742	5.3	12.6	546
3.0	5.7	6.6	522	5.6	5.8	386	5.7	19.8	770	5.4	16.0	560
3.5	5.9	8.3	536	5.6	7.2	388	5.8	28.4	810	5.6	17.8	564
4.0	6.1	9.8	560	5.8	8.4	396	6.0	31.6	786	5.6	21.0	574
4.5	6.2	11.8	576	6.0	11.6	428	6.2	39.6	818	5.8	24.4	580
5.0	6.4	13.1	570	6.0	11.8	410	6.1	44.4	838	5.8	27.2	584
5.5	6.5	14.6	592	6.2	13.8	412	6.3	50.6	872	6.0	33.2	594
6.0	6.6	18.8	600	6.3	16.0	410	6.4	54.4	888	6.0	41.1	610
7.0	6.7	18.0	582	6.3	19.4	416	6.3	64.0	916	6.0	43.0	612
8.0	6.7	18.8	610	6.5	22.3	422	6.5	70.6	948	6.1	48.2	628
24	6.9	20.5	638	6.6	29.4	430	6.8	84.6	1,054	6.3	61.7	636
72	7.3	21.2	708	7.1	39.1	426	6.9	64.4	1,108	6.4	77.8	690
168	8.2	19.7	742	7.2	41.9	426	7.3	41.6	1,166	6.4	93.6	722
264	8.5	17.6	772	7.2	41.3	438	8.2	28.9	1,244	6.5	90.8	766
336	8.6	18.6	778	7.4	45.5	436	8.5	25.8	1,270	6.5	108	788

a. Conducted at U.S. Geological Survey, New Cumberland, Pennsylvania, laboratory May 3-17, 2004, following methods of Cravotta (2003). Each test used 2 kilograms of sieved limestone or steel-slag fragments and untreated effluent from the respective site. Composition of steel slag and limestone indicated in tables 4 and 5. Data for elapsed time = 0 represented by samples for sites 1 and Fe that were collected April 27, 2004 (table 2).

b. Concentrations and pH measured after filtration through 0.45-micrometer pore-size filter.

c. Calcium concentration determined by colorimetric titration (American Public Health Association, 1992a); alkalinity determined by electrometric titration to pH 4.5 endpoint (American Public Health Association, 1998b).

Table 7. Cubitainer[a] test data for specific conductance, pH, alkalinity, computed acidity, and corresponding concentrations of dissolved[b] major cations and anions in effluent from reaction between steel slag or limestone with abandoned mine drainage (AMD) from Sites 1 and Fe at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania

[hr, hours; <, less than; >, greater than; µS/cm, microsiemens per centimeter; mg/L, milligrams per liter; n.d., no data]

Elapsed Time	Specific Conductance	pH	Alkalinity	Acidity computed	Net Acidity computed	Aluminum as Al	Iron as Fe	Manganese as Mn	Calcium as Ca[c]	Calcium as $CaCO_3$	Magnesium as Mg	Sodium as Na	Potassium as K	Silica as SiO_2	Sulfate as SO_4	Nitrate as N	Nitrite as N	Phosphate as P	Chloride as Cl	Fluoride as F	Bromine as Br	Iodide as I
(hr)	(µS/cm)	(units)	(mg/L as $CaCO_3$)			(mg/L)			(mg/L)						(mg/L)							
Site 1 AMD + Steel Slag																						
0	990	3.7	0.0	56	56.0	4.96	0.96	1.48	121	303	47.6	4.08	1.47	18.8	612	0.53	<0.04	<0.08	10.8	0.16	<0.030	0.023
1	950	5.1	4.2	20	15.8	3.05	<.10	1.41	197	493	50.8	5.03	1.67	24.2	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	<.030	<.010
8	1,200	6.7	18.8	2	-16.8	.10	.12	.92	>200	>500	46.5	5.11	1.76	33.2	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	<.030	<.010
336	1,700	8.6	18.6	1	-17.6	<.02	.15	.24	>200	>500	35.2	4.52	1.53	57.8	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	<.030	.033
Site 1 AMD + Limestone																						
0	990	3.7	0.0	56	56.0	4.96	0.96	1.48	121	303	47.6	4.08	1.47	18.8	612	0.53	<0.04	<0.08	10.8	0.16	<0.030	0.023
1	900	5.0	3.1	25	21.9	3.94	.18	1.51	147	368	52.2	4.85	1.68	19.0	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	<.030	<.010
8	1,250	6.5	22.3	3	-19.3	.08	<.10	1.46	169	423	53.7	4.74	1.68	17.1	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	<.030	<.010
336	1,790	7.4	45.5	2	-43.5	<.02	.16	1.12	175	438	51.8	4.89	1.73	14.0	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	<.030	<.010
Site Fe AMD + Steel Slag																						
0	1,610	4.0	0.0	194	194.0	9.31	71.3	4.40	191	478	71.6	4.29	2.14	31.0	1080	<0.05	<0.05	<0.1	18.7	0.44	<0.030	0.020
1	1,600	5.0	7.2	169	161.8	7.21	64.1	7.22	>200	>500	80.0	5.22	2.36	34.2	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.045	<.010
8	1,900	6.5	70.6	64	-6.6	.32	23.1	11.7	>200	>500	74.6	5.20	2.37	35.3	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.037	<.010
336	2,230	8.5	25.8	2	-23.8	<.02	.22	.99	>200	>500	52.4	4.11	1.77	58.0	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.043	.017
Site Fe AMD + Limestone																						
0	1,610	4.0	0.0	194	194.0	9.31	71.3	4.40	191	478	71.6	4.29	2.14	31.0	1080	<0.05	<0.05	<0.1	18.7	0.44	<0.030	0.020
1	1,610	4.9	8.8	183	174.2	7.14	74.8	4.60	>200	>500	82.1	4.91	2.46	32.9	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.043	<.010
8	1,700	6.1	48.2	140	91.8	.81	70.9	4.57	>200	>500	80.0	4.93	2.39	31.0	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.042	<.010
336	1,870	6.5	108	64	-44.0	<.02	31.6	4.14	>200	>500	80.3	4.95	2.36	23.1	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.n.d.	.034	<.010

a. Conducted at U.S. Geological Survey, New Cumberland, Pennsylvania, laboratory May 3-17, 2004, following methods of Cravotta (2003). Each test used 2 kilograms of sieved limestone or steel-slag fragments and untreated effluent from the respective site. Data for elapsed time = 0 represented by samples for sites 1 and Fe that were collected April 27, 2004 (tables 2 and 3).

b. Cations analyzed by inductively coupled plasma emission mass spectrometry and anions by ion chromatography (Crock and others, 1999) after 0.45-millimeter pore-size filtration.

c. Calcium concentration as $CaCO_3$ computed by multiplying Ca determined by inductively coupled plasma emission mass spectrometry by 2.5.

Table 8. Cubitainer[a] test data for dissolved trace elements[b] in effluent from reaction between steel slag or limestone with abandoned mine drainage (AMD) from Sites 1 and Fe at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania
[hr, hours; <, less than]

Elapsed Time (hr)	Element Concentration in Micrograms per Liter																											
	Ag	As	Au	Ba	Be	Bi	Cd	Ce	Co	Cr	Cs	Cu	Dy	Er	Eu	Ga	Gd	Ge	Hf	Hg	Ho	In	La	Li	Lu	Mo	Nb	Nd
Site 1 AMD + Steel Slag																												
0	<2	1.66	<0.02	12.5	1.97	<3	0.41	3.93	62	9.0	0.03	6.86	1.6	0.81	0.25	0.10	1.57	<.10	0.04	<2	0.31	<0.01	1.35	46	0.11	<1.0	<0.05	2.78
1	<2	3.06	<.02	36.7	1.29	<3	.40	3.99	50	29.9	.04	4.69	1.17	.64	.21	<.10	1.22	<.10	.03	<2	.23	<.01	2.05	52	.09	<1.0	<.05	2.58
8	<2	1.35	<.02	37.5	<1.00	<3	.17	.41	23	22.2	.05	<2.00	.07	.05	.02	<.10	.09	<.10	<.01	<2	.02	<.01	.34	49	<.01	<1.0	<.05	.24
336	<2	.78	.28	23.5	<1.00	<3	<.10	.02	4	12.0	.04	<2.00	<.01	<.01	<.01	<.10	<.01	<.10	<.01	<2	<.01	<.01	.02	52	<.01	<1.0	<.05	.01
Site 1 AMD + Limestone																												
0	<2	1.66	<0.02	12.5	1.97	<3	0.41	3.93	62	9.0	0.03	6.86	1.6	0.81	0.25	0.10	1.57	<0.10	0.04	<2	0.31	<0.01	1.35	46	0.11	<1.0	<0.05	2.78
1	<2	2.31	<.02	19.2	1.33	<3	.41	2.87	58	21.1	.01	5.53	1.08	.62	.19	<.10	1.07	<.10	.04	<2	.23	<.01	1.06	48	.07	<1.0	<.05	1.85
8	<2	1.36	<.02	27.9	<1.00	<3	.32	.66	58	20.3	.01	<2.00	.19	.09	.03	<.10	.20	<.10	<.01	<2	.04	<.01	.33	47	.01	<1.0	<.05	.38
336	<2	<.30	<.02	43.1	<1.00	<3	.17	.09	43	9.9	.01	<2.00	<.01	<.01	<.01	<.10	.01	<.10	<.01	<2	<.01	<.01	.07	43	<.01	<1.0	<.05	.04
Site Fe AMD + Steel Slag																												
0	<2	0.36	<0.02	7.15	7.86	<3	0.54	13.1	159	<5.0	0.07	<2.00	7.25	3.71	1.18	0.20	6.88	0.11	0.14	<2	1.39	<0.01	3.46	107	0.47	<1.0	<0.05	10.2
1	<2	<.30	<.02	37.3	6.07	<3	.55	14.5	144	16.8	.08	2.43	5.3	2.88	.90	.20	5.37	.14	.11	<2	1.02	<.01	4.28	122	.36	<1.0	<.05	8.52
8	<2	<.30	<.02	55.7	<1.00	<3	.24	4.51	108	11.7	.08	<2.00	.57	.32	.10	.24	.64	<.10	.02	<2	.12	<.01	1.95	130	.04	<1.0	<.05	1.26
336	<2	1.83	<.02	22.3	<1.00	<3	<.10	.02	4	16.6	.06	<2.00	<.01	<.01	<.01	<.10	<.01	<.10	<.01	<2	<.01	<.01	.01	131	<.01	1.1	<.05	<.01
Site Fe AMD + Limestone																												
0	<2	0.36	<0.02	7.15	7.86	<3	0.54	13.1	159	<5.0	0.07	<2.00	7.25	3.71	1.18	0.20	6.88	0.11	0.14	<2	1.39	<0.01	3.46	107	0.47	<1.0	<0.05	10.2
1	<2	<.30	<.02	16.9	7.12	<3	.67	10.1	160	13.9	.06	3.39	5.41	2.74	.88	.14	5.19	.12	.12	<2	1.04	<.01	3.05	122	.34	<1.0	<.05	7.83
8	<2	<.30	<.02	29.8	2.30	<3	.62	6.02	158	12.1	.05	<2.00	2.15	1.15	.33	.12	2.34	<.10	.05	<2	.45	<.01	2.24	119	.11	<1.0	<.05	3.56
336	<2	<.30	<.02	39.5	<1.00	<3	.34	.19	144	9.0	.04	<2.00	.08	.06	<.01	<.10	.06	<.10	<.01	<2	.02	<.01	.19	119	<.01	<1.0	<.05	.07

a. Conducted at U.S. Geological Survey, New Cumberland, Pennsylvania, laboratory May 3-17, 2004, following methods of Cravotta (2003). Each test used 2 kilograms of sieved limestone or steel-slag fragments and untreated effluent from the respective site.

b. Concentration determined by inductively coupled plasma emission mass spectrometry (Crock and others, 1999) after 0.45-millimeter pore-size filtration.

Table 8. Cubitainer[a] test data for dissolved trace elements[b] in effluent from reaction between steel slag or limestone with abandoned mine drainage (AMD) from Sites 1 and Fe at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania (continued)
[hr, hours; <, less than]

Elapsed Time (hr)	Element Concentration in Micrograms per Liter																												
	Ni	Os	Pb	Pd	Pr	Pt	Rb	Re	Ru	Sb	Sc	Se	Sm	Sn	Sr	Ta	Tb	Te	Th	Ti	Tl	Tm	U	V	W	Y	Yb	Zn	Zr
Site 1 AMD + Steel Slag																													
0	107	<0.02	0.84	<0.10	0.59	<3	2.30	<0.01	<0.1	<0.1	<10	7.14	0.91	<1	204	<.01	0.29	<1	0.02	<1.00	0.48	0.11	0.49	2.44	<0.2	7.59	0.70	200	<0.1
1	99	<.02	.10	<.10	.61	<3	5.91	<.01	<.1	<.1	<10	14.1	.76	<1	294	<.01	.21	<1	<.01	1.72	.21	.08	.55	9.07	<.2	6.43	.56	161	<.1
8	62	<.02	.02	<.10	.05	<3	8.30	<.01	<.1	<.1	<10	7.44	.05	<1	360	<.01	.01	<1	<.01	2.45	.03	<.01	.37	6.80	<.2	.53	.03	33	<.1
336	19	<.02	<.10	.69	<.01	<3	8.33	<.01	<.1	<.1	<10	6.97	<.01	<1	524	<.01	<.01	<1	<.01	4.33	<.01	<.01	.37	5.34	<.2	.08	<.01	23	<.1
Site 1 AMD + Limestone																													
0	107	<0.02	0.84	<0.10	0.59	<3	2.30	<0.01	<0.1	<0.1	<10	7.14	0.91	<1	204	<0.01	.29	<1	0.02	<1.00	0.48	0.11	0.49	2.44	<0.2	7.59	0.70	200	<0.1
1	103	<.02	.93	<.10	.41	<3	2.05	<.01	<.1	<.1	<10	8.20	.65	<1	229	<.01	.20	<1	<.01	<1.00	.42	.08	.43	6.11	<.2	5.75	.48	195	<.1
8	106	<.02	<.10	<.10	.08	<3	1.68	<.01	<.1	<.1	<10	4.30	.10	<1	270	<.01	.04	<1	<.01	<1.00	.33	.01	.08	5.95	<.2	1.18	.09	145	<.1
336	75	<.02	<.10	<.10	<.01	<3	1.23	<.01	<.1	<.1	<10	<2.00	<.01	<1	404	<.01	<.01	<1	<.01	<1.00	.16	<.01	.42	2.89	<.2	.13	<.01	80	<.1
Site Fe AMD + Steel Slag																													
0	247	<0.02	1.37	<0.10	2.02	<3	5.73	<0.01	<0.1	<0.1	<10	<2.00	4.14	<1	407	0.02	1.30	<1	<0.01	2.10	0.22	0.47	0.50	1.92	<0.2	32.6	3.03	574	<0.1
1	234	<.02	.51	<.10	1.73	<3	10.7	<.01	<.1	<.1	<10	<2.00	3.22	<1	524	.01	.97	<1	<.01	3.03	.10	.37	.75	4.77	<.2	26.6	2.32	504	<.1
8	186	<.02	<.10	<.10	.31	<3	14.8	<.01	<.1	<.1	<10	2.79	.33	<1	675	<.01	.10	<1	<.01	2.94	.01	.03	2.39	3.66	<.2	4.54	.21	208	<.1
336	18	<.02	<.10	.72	<.01	<3	10.6	<.01	<.1	<.1	<10	8.85	<.01	<1	851	<.01	<.01	<1	<.01	4.28	<.01	<.01	1.42	5.67	<.2	<.03	<.01	8	<.1
Site Fe AMD + Limestone																													
0	247	<0.02	1.37	<0.10	2.02	<3	5.73	<0.01	<0.1	<0.1	<10	<2.00	4.14	<1	407	0.02	1.30	<1	<0.01	2.10	0.22	0.47	0.50	1.92	<0.2	32.6	3.03	574	<0.1
1	250	<.02	1.08	<.10	1.55	<3	5.57	<.01	<.1	<.1	<10	<2.00	3.08	<1	452	.01	.96	<1	<.01	2.32	.19	.36	.39	4.12	<.2	27.0	2.22	608	<.1
8	255	<.02	<.10	<.10	.77	<3	5.01	<.01	<.1	<.1	<10	<2.00	1.15	<1	514	<.01	.40	<1	<.01	2.63	.16	.14	.10	3.51	<.2	14.6	.81	564	<.1
336	227	<.02	<.10	<.10	.02	<3	3.67	<.01	<.1	<.1	<10	<2.00	.01	<1	697	<.01	.01	<1	<.01	1.91	.08	<.01	.20	2.55	<.2	1.88	.05	416	<.1

a. Conducted at USGS New Cumberland, Pennsylvania, laboratory May 3-17, 2004, following methods of Cravotta (2003). Each test used 2 kilograms of sieved limestone or steel-slag fragments and untreated effluent from the respective site.

b. Concentration determined by inductively coupled plasma emission mass spectrometry (Crock and others, 1999) after 0.45-millimeter pore-size filtration.

Table 9. Summary of cubitainer test conditions and results for rate of reaction between steel slag or limestone with abandoned mine drainage (AMD) from Sites 1 and Fe at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania

[hr, hours; mg/L, milligrams per liter; L/mg/hr, liters per milligram per hour; cm^2/mg/hr, square centimeters per milligram per hour; cm^2/g, square centimeters per gram; IAP, ion activity product; Kt, equilibrium constant at temperature indicated; atm, atmospheres; <, less than; >, greater than]

Site/Test[a]	pH (units)					Calcite Saturation Index[b,c] (log IAP/Kt)					Carbon Dioxide Pressure[b,c] (log atm)					Alkalinity Rate Constants[d]						Calcium Rate Constants[d]					
																C_0	C_S	k' ($x10^2$)	K' ($x10^5$)	k'' ($x10^4$)	K'' ($x10^6$)	C_0	C_S	k' ($x10^2$)	K' ($x10^5$)	k'' ($x10^4$)	K'' ($x10^6$)
	Elapsed Time					Elapsed Time					Elapsed Time					(mg/L $CaCO_3$)		(1/hr)	(L/cm^2/hr)	(L/mg/hr)	(L^2/cm^2/mg/hr)	(mg/L $CaCO_3$)		(1/hr)	(L/cm^2/hr)	(L/mg/hr)	(L^2/cm^2/mg/hr)
	0 hr	1 hr	4 hr	8 hr	336 hr	0 hr	1 hr	4 hr	8 hr	336 hr	0 hr	1 hr	4 hr	8 hr	336 hr												
Site 1 AMD + Steel Slag	3.7	5.1	6.1	6.7	8.6	<-4.0	-3.7	-2.2	-1.3	0.7	>-1.3	-1.5	-2.2	-2.4	-4.4	0.0	21.2	-27.66	-31.72	389.54	44.67	312	778	-11.89	-13.64	4.57	0.52
Site 1 AMD + Limestone	3.7	5.0	5.8	6.3	7.2	<-4.0	-4.0	-2.7	-2.0	-.6	>-1.5	-1.5	-1.9	-2.1	-2.6	.0	45.5	-7.51	-9.01	21.21	2.55	312	438	-24.45	-29.35	64.31	7.72
Site Fe AMD + Steel Slag	4.0	5.0	6.0	6.5	8.5	<-3.9	-3.4	-1.7	-.8	.8	>-1.2	-1.2	-1.6	-1.7	-4.1	.0	84.6	-19.13	-21.94	43.43	4.98	480	1270	-10.15	-11.64	2.03	0.23
Site Fe AMD + Limestone	4.0	4.9	5.6	6.1	6.5	<-3.9	-3.6	-2.4	-1.5	-.8	>-1.1	-1.0	-1.4	-1.5	-1.6	.0	107.6	-6.98	-8.37	8.37	1.00	480	788	-10.25	-12.30	4.23	0.51

a. All tests conducted in laboratory May 3-17, 2004, using collapsible 1-gallon nominal polyethylene containers filled with fresh influent from the discharge and 2 kilograms sieved fragments (0.5 to 1.5 inches) of limestone or steel slag with effluent recirculated under closed conditions. Cubitainer test data used for rate estimates are shown in figure 5 and table 6. Data on composition of steel slag and limestone are shown in tables 4 and 5.

b. Calcite saturation index (SI) and partial pressure of carbon dioxide (Pco_2) computed using measured temperature, specific conductance (SC), pH, alkalinity as HCO_3^-, and calcium as Ca (Langmuir, 1997; American Public Health Association, 1992b); van't Hoff temperature-corrected equilibrium constants from Ball and Nordstrom (1991); and Debye-Huckel activity coefficients on the basis of estimated ionic strength, $I = 1.88 \cdot (10^{-5}) \cdot SC$ (Langmuir, 1997). SC values are shown in table 7.

c. Alkalinity value greater than 0 is needed to compute SI and Pco_2. Numerical values for SI and Pco_2 at 0 h are given for first sample (at 0.5 or 1.0 h) with positive alkalinity.

d. First-order rate equation, $C_t = C_S - [(C_S - C_0) \cdot \exp\{-k' \cdot t_d\}]$, or second-order rate equation, $C_t = C_S - \{1/[k'' \cdot t_d + 1/(C_S - C_0)]\}$, used to estimate alkalinity or calcium concentration as a function of detention time on the basis of the initial and steady-state concentration (C_0 and C_S) and the overall first- and second-order rate constants (k' and k'') derived from cubitainer tests. Normalized rate constant (K' or K'') was computed as the product of the overall rate constant and the initial solution volume divided by the surface area of substrate in the cubitainer: $K = k \cdot (V/A)$, where the total surface area was estimated as the product of the mass and unit surface area of the solid substrate. The unit surface areas of the limestone and steel-slag substrates used in the cubitainer tests were estimated on the basis of the measured dimensions, weight, and volume of 50 samples of each substrate. Assuming an ellipsoidal sphere as the geometry, the average unit surface area of limestone and steel slag was 1.53 and 1.52 cm^2/g, respectively (appendix C). The overall rate constant for other particle sizes can be computed as the product of the normalized rate constant and the surface area of that particle divided by the solution volume. Because alkalinity values initially were less than 0 but were positive after 0.5 hr detention time, a time lag of 0.4 hr was assumed for the estimated alkalinity concentration. Substituting $t_d = (t_d - 0.4)$ in the rate equations yields alkalinity values less than zero for detention times less than 0.4 hr, and alkalinities greater than 0 for detention times greater than 0.4 hr.

Table 10. Rankings of pollutant loading and possible remedial alternatives for abandoned mine drainage at the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site, Pennsylvania
[Priority rankings based on instantaneous loadings of dissolved iron, aluminum, and manganese during April 7 and 27, 2004. Remedial alternatives are not identified in order of preference; any treatment design would require additional data and specific analysis; VFCW, vertical-flow compost wetland; ALD, anoxic limestone drain; OLD, flushable oxic limestone drain; OLC, open limestone channel; X, applicable; +, additional; - not applicable]

Local Site Name	Pollutant Loading Rank[a]	Principal Characteristics[b]	Remedial Alternatives[c]						
			Remine/Reclaim	VFCW	ALD	OLD	OLC	Aerobic Pond(s)	Active Treatment
U	4	Moderate flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	X
1	1	Moderate flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	X
1Fe	10	Small flow; very high Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	-
Fe	12	Small flow; very high Fe, Mn; very high Al; net acidic; suboxic	X	X	-	-	-	+	-
2	7	Small flow; high Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	-
3	6	Moderate flow; low Fe, Mn; moderate Al; net acidic; oxic	-	-	-	X	X	+	-
3A	11	Small flow; low Fe, Mn; high Al; net acidic; oxic	X	X	-	X	X	+	-
3B	8	Moderate flow; moderate Fe, Mn; high Al; net acidic; oxic	X	X	-	X	X	+	-
4	9	Small flow; low Fe, Mn; very high Al; net acidic; oxic	-	X	-	-	-	+	-
5	5	Moderate flow; low Fe, Mn; moderate Al; net acidic?; oxic	-	X	-	X	X	+	-
6	3	Moderate flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	X
7	2	Moderate flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	X
1+1Fe+Fe+2[d]	n.a.	Moderate flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	X
3+3B+5+6+7	n.a.	Very large flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	X
1+1Fe+Fe+2+ 3+3B+5+6+7	n.a.	Very large flow; moderate Fe, Mn; very high Al; net acidic; oxic	X	X	-	-	-	+	X

a. Rankings based on loading computed as product of maximum flow rate and maximum concentrations of dissolved iron, aluminum, and manganese for samples collected on April 7 and 27, 2004 (table 2). Rank of 1 for greatest loading; rank value increases with decreased loading.

b. Principal characteristics based on maxima and minima for flow rate and concentrations of alkalinity, dissolved metals, and oxygen (in mg/L) for samples collected on April 7 and 27, 2004 (table 2). Flow (gal/min): "very large" if minimum > 1,000; "large" if maximum ≥ 500 and < 1,000; "moderate" if maximum ≥ 50 and < 500; "small" if maximum < 50; "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 and ≤ 4; "moderate" if maximum > 0.2 and ≤ 2; "low" if maximum < 0.2. Net alkalinity (alkalinity - computed acidity; mg/L as $CaCO_3$): "net acidic?" if maximum ≤ 5; "net acidic" if maximum ≤ 0; "net alkaline?" if minimum > 0 or no data 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.

c. Remedial alternatives initially identified on the basis of maxima and minima for flow rate (in gal/min) and water quality (in mg/L): "Remine/reclaim" 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 ≤ 300; "ALD and aerobic pond" if minimum net alkalinity < 5, maximum dissolved oxygen ≤ 1, maximum Al < 2, and maximum flow ≤ 300; "OLD and aerobic pond" if minimum net alkalinity < 5, maximum dissolved oxygen > 1, maximum Al < 3, and maximum flow ≤ 300; "OLC" if minimum net alkalinity < 5, maximum Fe < 10, maximum Al < 5, and maximum flow ≤ 4,500; "Active Treatment" if minimum flow > 2 or maximum net alkalinity < -300. The option to "Remine/reclaim" with respect to the ALPO-SBTU AMD sources would require this activity outside of the ALPO-SBTU boundaries and, thus, is beyond the authority or responsibility of the National Park Service.

d. AMD sites were combined considering possible flow paths and potential for merging or aggregation of AMD. The cumulative flow rate and flow-weighted average concentrations were used to estimate combined AMD characteristics. The combinations exclude sites U, 3A, and 4 because these are above sites 1, 3B, and 5, respectively. Site 8 was excluded because, on the basis of its chemistry, it is not AMD.