Swatara National Monitoring Project

Final Report

Contract # 3521671

Sponsor: Schuylkill County Conservation District

Executive Summary

Swatara Creek is located eastern Pennsylvania, HUC 02050305**.** The headwater portion of the watershed is affected by abandoned mine drainage (AMD). Much of Swatara Creek above Ravine and many tributaries including Panther Creek, Coal Run, Middle Creek, Good Spring Creek, Lower Rausch Creek and Lorberry Creek are impaired due to metals, pH, siltation and suspended solids from AMD. In April 1999, EPA approved the Upper Swatara Creek TMDL which addresses Swatara Creek and all the streams mentioned above.

Most of the AMD contaminating Swatara Creek is from legacy anthracite mines. This contamination resulted in poor water quality and little or not aquatic life present in this area. A variety of passive and semi-passive treatment systems (along with some land reclamation activities) were implemented to neutralize acidic mine drainage (AMD) and reduce the transport of dissolved metals in the upper Swatara Creek Basin in the Southern Anthracite Coalfield in eastern Pennsylvania. To evaluate the effectiveness of selected treatment systems installed during 1995-2001, water-quality data were collected at upstream and downstream locations relative to each system eight or more times annually during 1996-2007. Performance was normalized among treatment types by dividing the acid load removed by the size of the treatment system. For the limestone sand, open limestone channel, oxic limestone drain, anoxic limestone drain, and limestone diversion well treatment systems, the size was indicated by the total mass of limestone; for the aerobic wetland systems, the size was indicated by the total surface area of ponds and wetlands. Additionally, the approximate cost per ton of acid treated over an assumed service life of 20 years was computed. On the basis of these performance metrics, the limestone sand, anoxic limestone drain, oxic limestone drain, and limestone diversion wells had similar ranges of acid-removal efficiency and cost efficiency. However, the open limestone channel had lower removal efficiency and higher cost per ton of acid treated. The wetlands effectively attenuated metals transport but were relatively expensive considering metrics that evaluated acid removal and cost efficiency. Although the water-quality data indicated that all treatments reduced the acidity load from AMD, the anoxic limestone drain was most effective at producing near-neutral pH and attenuating acidity and dissolved metals. The diversion wells were effective at removing acidity and increasing pH of downstream water and exhibited unique potential to treat moderate to high flows associated with stormflow conditions.

Aquatic life was also sampled to gauge the effects of the various measures taken to treat AMD. Intermittently collected base-flow data for 1959-1986 indicate that fish were absent immediately downstream from the mined area where pH ranged from 3.5 to 7.2 and concentrations of sulfate, dissolved iron, and dissolved aluminum were as high as 250, 2.0, and 4.7 mg/L, respectively. However, in the 1990s, fish returned to upper Swatara Creek, coinciding with the implementation of AMD treatments (listed above) in the watershed. During 1996-2006, as many as 25 species of fish were identified in the reach downstream from the mined area with base-flow pH from 5.8 to 7.6 and concentrations of sulfate, dissolved iron, and dissolved aluminum as high as 120, 1.2, and 0.43 mg/L, respectively. Several of the fish taxa were intolerant of pollution and low pH, such as river chub (*Nocomis micropogon*) and longnose dace (*Rhinichthys cataractae*). Cold-water species such as brook trout (*Salvelinus fontinalis*) and warm-water species such as rock bass (*Ambloplites rupestris*) varied in predominance depending on streamflow and stream temperature.

Stormflow data for 1996-2007 indicated pH, alkalinity, and sulfate concentrations decreased as the streamflow and associated storm-runoff component increased, whereas iron and other metal concentrations were poorly correlated with streamflow because of hysteresis effects (greater metal concentrations during rising stage than falling stage). Prior to 1999, pH < 5.0 was recorded during several storm events; however, since the implementation of AMD treatments, pH has been maintained near neutral. Flow-adjusted trends for 1997-2006 indicated significant increases in calcium; decreases in hydrogen ion, dissolved aluminum, dissolved and total manganese, and total iron; and no change in sulfate or dissolved iron in Swatara Creek immediately downstream

from the mined area. The increased pH and calcium from limestone in treatment systems can be important for regulating toxic effects of dissolved metals. Thus, treatment of AMD during the 1990s improved pH buffering, reduced metals transport, and helped to decrease metals toxicity to fish.

The benthic macroinvertebrate community was sampled earlier in the study. The last year this sampling occurred was in 1999. The benthic macroinvertebrate community sampled at Ravine did not show the same increase as the fish community. The calculated Hilsenhoff's (1988) family biotic index indicated improved water quality, just not as dramatic as the fish.

Introduction

"Acidic" mine drainage (AMD) commonly has acidic pH (< 4.5) and elevated concentrations of dissolved and particulate iron (Fe) and dissolved sulfate (SO_4^{2-}) that result from the oxidation of pyrite (FeS_2) in coal-bearing rock (Rose and Cravotta 1998). Half the proton acidity (H^+) produced by the stoichiometric oxidation of FeS_2 results from the oxidation of pyritic sulfur to SO_4^{2-} and the other half results from the oxidation of ferrous (Fe^{2+}) to ferric (Fe^{3+}) iron and its consequent precipitation as $Fe(OH)_3$ and related solids (Bigham and Nordstrom 2000; Cravotta et al. 1999). Because AMD commonly contains Fe^{2+} when discharged at the land surface, the pH of receiving streamwater may decline as the water becomes oxygenated and oxidation and hydrolysis reactions proceed (e.g. Cravotta and Kirby 2004; Kirby and Cravotta 2005). Dissolved concentrations of sulfate (SO_4^{2-}), iron (Fe^{2+} and Fe^{3+}), manganese (Mn^{2+}), aluminum (Al^{3+}), zinc (Zn^{2+}), nickel (Ni^{2+}), copper (Cu^{2+}), lead (Pb^{2+}) and other solutes commonly are elevated in AMD due to aggressive dissolution of aluminosilicate, oxide, and carbonate minerals by the low-pH water (Blowes et al. 2003; Cravotta 1994, 2008).

The acid produced by pyrite oxidation and by hydrolysis of dissolved Fe^{2+}, Fe^{3+}, and other cations can be neutralized by reaction with calcite ($CaCO_3$) and dolomite [$CaMg(CO_3)_2$]. These calcareous minerals are the dominant components of limestone and can occur in nodules, cementing agents, or fractures in sandstone, siltstone, shale, and associated strata of coal-bearing rocks. Where absent or deficient at a mine site, the addition of limestone or other alkalinity-producing materials to mine spoil or mine drainage can be effective for prevention or neutralization of AMD. Alkalinity, represented by bicarbonate (HCO_3^-), and base cations including calcium (Ca^{2+}) and magnesium (Mg^{2+}) are common products of neutralization by limestone.

The transport of dissolved Fe, Al, Mn, and other metals from AMD sources typically is attenuated owing to precipitation and adsorption processes, which can vary as a function of pH or redox state. Under anoxic conditions in flooded underground mines, concentrations of Fe^{2+} and Mn^{2+} can remain elevated owing to relatively high solubility of Fe^{II} and Mn^{II} oxyhydroxides and carbonates (Cravotta 2008a). However, under oxidizing conditions at the surface, the attenuation of dissolved cations generally increases as pH approaches neutrality (pH 6-8). At pH >3, concentrations of Fe^{3+} tend to be limited by the formation of Fe^{III}-oxyhydroxides, and at pH > 5, concentrations of Al^{3+} and, to a lesser extent, Mn^{2+} tend to be limited by the formation of Al and Mn^{III-IV} oxyhydroxide compounds, respectively (Bigham and Nordstrom 2000; Cravotta 2008; Rose and Cravotta 1998). These oxyhydroxides can be effective adsorbents of dissolved cations and anions in AMD (Kairies et al. 2005; Webster et al. 1998).

Where reclamation of a mine and prevention of AMD are not feasible, treatment of the AMD may be warranted to neutralize acidity and remove dissolved and suspended pollutants from the aquatic system. Generally, if the AMD has excess alkalinity (net acidity < 0; hot acidity < 0), the pH of the AMD will be maintained near neutral after atmospheric equilibration (Kirby and Cravotta 2005), and oxidation ponds or aerobic wetlands can be useful to remove precipitated metals (Hedin et al.1994a). However, if the AMD has deficient alkalinity (net acidity > 0; hot acidity > 0), a supplemental alkalinity source is needed to maintain near- neutral pH. Conventional "active" treatment of AMD involves the addition of caustic chemicals, such as sodium hydroxide (NaOH) or hydrated lime ($Ca(OH)_2$), to increase pH and remove dissolved metals (Skousen et al. 1998). Alternatively, "passive" and "semi-passive" AMD treatment systems can be used that include anaerobic and aerobic wetlands and various limestone-based systems, such as anoxic or oxic limestone drains, open limestone channels, limestone diversion wells, and vertical flow compost wetlands (Hedin et al. 1994a; Skousen et al. 1998; Watzlaf et al. 2004; Ziemkiewicz et al. 2003). These passive and semi-passive systems generally are limited by slower rates of neutralization and pollutant removal than active treatments but can be cost effective where water chemistry meets suggested criteria and where land and component materials are locally available (Ziemkiewicz et al. 2003). If direct treatment of the AMD is not feasible, pH adjustment of the streamwater may be effective to meet water-quality goals.

Various passive- and semi-passive treatment systems have different advantages and disadvantages; however, all suffer from possible complications associated with variability of flow rates and chemistry of the AMD and from uncertainties about efficiency and longevity of the treatment. Furthermore, every site requiring treatment has unique environmental characteristics. In general, passive-treatment systems are effective for treating the average or "normal" water-quality conditions (Skousen et al. 1998; Ziemkiewicz et al. 2003). Nevertheless, treatment effectiveness and downstream benefits could be diminished as conditions deviate from normal. For example, the performance of a treatment system could decline with increased flow rate because of decreased retention time and increased contaminant loading. However, treatment performance generally is poorly characterized for a wide range of flow conditions.

Drainage from abandoned mines affects the water quality and aquatic ecology of streams and lakes in coal and metal mining regions worldwide (Nordstrom 2000; Wolkersdorfer and Bowell 2004, 2005a, 2005b). For example, legacy mining in the Appalachian Coalfield of the eastern USA has transformed the local landscape and rendered many streams fishless because of "acidic" mine drainage (AMD) (Herlihy et al. 1990; U.S. Environmental Protection Agency 1995). In Pennsylvania, AMD from abandoned coal mines is the leading cause of nonpoint-source (NPS) pollution, degrading approximately 8,800 km of streams (Pennsylvania Department of Environmental Protection 2004, 2007) and accounting for lost revenues of approximately $67 million annually because of recreational fishing losses (Pennsylvania Organization for Watersheds and Rivers 2002).

AMD reactions are complex and the effects can be dramatic to aquatic life in a stream. Low pH and elevated concentrations of dissolved metals in the water column and pore water of stream sediment can be stressful or toxic to fish and aquatic macroinvertebrates (Baker and Schofield 1982; Burrows 1977; Butler et al. 1973; Courtney and Clements 2002; Dsa et al. 2008; MacDonald et al. 2000; U.S. Environmental Protection Agency 2002). The transport of dissolved metals across biological membranes and/or ingestion of contaminated food or sediment with subsequent transport across the gut are the primary routes of toxic exposure (Elder 1988; Havas and Rosseland 1995). Additionally, dissolved Al^{3+} and Fe^{3+} can precipitate on the gills or equivalent organs, suffocating aquatic organisms (Cleveland et al. 1991; Havas and Rosseland 1995; Henry et al. 1999).

The severity of metals toxicity tends to be greater under low-pH conditions than under near-neutral conditions. Accordingly, the U.S. Environmental Protection Agency (2002) recommends pH 6.5 to 9.0 for protection of freshwater aquatic life, and the Commonwealth of Pennsylvania (2002) stipulates that effluent discharged from active mines must have pH 6.0 to 9.0 *and* alkalinity greater than acidity. Near- neutral pH could result from dissolution of limestone and other calcareous bedrock by the AMD (e.g. Cravotta et al. 1999) or from mixing of acidic AMD with neutral, carbonate-buffered surface water (e.g. Broshears et al. 1996; Caruso 2005; Henry et al. 1999; Schemel et al. 2000). At near-neutral pH, concentrations of dissolved Al^{3+} and Fe^{3+} are limited by the precipitation of hydrous oxide and hydroxysulfate minerals, and the transport of other toxic metals, such as Cu^{2+}, Pb^{2+}, Ni^{2+}, and Zn^{2+}, typically is attenuated owing to adsorption to such minerals (Bigham and Nordstrom 2000; Coston et al. 1995; Cravotta 2008; Webster et al. 1998; Winland et al. 1991). Nevertheless, even if concentrations of solutes in the water column are below toxicity thresholds, the accumulation of metal-rich solids within the streambed can degrade the benthic habitat and affect trophic structure and reproduction (Cannon and Kimmel 1992; Dsa et al. 2008; Earle and Callaghan 1998; Havas and Rosseland 1995). Accordingly, strategies to treat the AMD before it discharges to streams commonly implement steps that increase pH and alkalinity, promote the oxidation of Fe^{2+} and Mn^{2+}, and facilitate the precipitation and settling of hydrous oxides of Fe^{III}, Mn^{III-IV}, Al, and other metal-rich compounds (Johnson and Hallberg 2005; Skousen et al. 1998; Watzlaf et al. 2004).

Chemical conditions in streams may rebound quickly following neutralization of AMD; however, the recovery of aquatic invertebrates, zooplankton, and fish may take decades (Chadwick and Canton 1986; Galloway 2001; Herricks 1977; Monteith et al. 2005; Vrba et al. 2003; Youndt and Niemi 1990). Instead of continuous accrual of species over the improving chemical gradient, recovery tends to be punctuated, with groups of taxa added as particular chemical thresholds are attained (Monteith et al. 2005). Impediments to ecological recovery of acidified systems include inadequate or unstable water quality, residual effects of degraded substrate or habitat, inadequate or inaccessible supply of organisms for recolonization, and community-level competition and dynamics (Findlay 2003; Herricks 1977; Nelson and Roline 1996; Short et al. 1990; Yan et al. 2003).

Despite historical degradation from AMD, reproducing populations of brook trout (*Salvelinus fontinalis*) and other native fishes recently have been documented in several streams in the Anthracite Coalfield of eastern Pennsylvania (Cravotta 2005; Cravotta and Bilger 2001; Cravotta and Kirby 2004; Cravotta and Nantz 2008) that had been considered fishless in 1995 (U.S. Environmental Protection Agency 1995). The recent appearance of fish coincides with improved water quality of the streams and associated AMD sources, characterized by near-neutral pH and decreased concentrations of dissolved metals and acidity (e.g. Jackson 1987; Wood 1996).

Purpose and Scope

The upper Swatara Creek has been the focus of numerous monitoring efforts and identification of mining related impacts for decades (since the 1950's). The previous monitoring efforts were to evaluate Swatara Creek for the construction of a water impoundment that would serve as an alternate drinking water source for the City of Lebanon and a recreation lake within Swatara State Park located 15 miles south of the anthracite mining area. The concept for the lake was developed in the 1960's. Early studies identified acid mine drainage a major pollutant. There were numerous projects completed by the Department of Environmental Resources, Bureau of Abandoned Mine Reclamation as part of project Scarlift tin the 1970's that targeted reclamation of abandoned mine and the construction of concrete flumes to convey streams that were previously lost to underground mine workings. Studies in the mid-1980's acknowledged that improvements were made in water quality, however there were outstanding sources of acid mine drainage pollution that impacted Swatara Creek and recommended they would need to be remediated prior to construction of a water impoundment. With limited resources the state began efforts to abate the mine drainage pollution in the early 1990's with a main objective of improving the water quality to acceptable standards which would allow the recreational lake at Swatara State Park to be constructed. As funding for mine drainage BMPs became more available in the mid-1990's the effort gained local support and the goal had been modified to restoring Swatara Creek to a viable fishery. According to the PA Fish and Boat Commission, the water quality necessary to establish a healthy ecosystem would be pH 6.0-6.5, alkalinity>acidity by 20 mg/l, iron <0.5 mg/l, and aluminum <0.5 mg/l.

As part of the emerging effort to restore Swatara Creek it was evident that monitoring was a critical component to document improvements, evaluate installed treatment systems, and to further direct the treatment efforts. Although there are over 40 sources of AMD in the Swatara Creek watershed all of them are in the headwaters. Swatara's headwaters are in the southern coal field of the Anthracite Region. Swatara Creek and its tributaries that are impacted with AMD, meet and flow through Second Mountain near the village of Ravine. The NNSPMP monitoring station was established on Swatara Creek at Ravine to serve as the primary monitoring station to evaluate cumulative effects of the AMD impacts as well as the marker for improvements made in the watershed.

Therefore, the purpose of this National Monitoring project was two-fold. First was to evaluate the effectiveness of passive and semi-passive treatment systems for neutralizing acidity and removing metals and other pollutants from AMD and affected streamwater. Data collected by the U.S. Geological Survey (USGS) over a wide range of flow conditions during June 1996 through June 2007 in the upper Swatara Creek Basin at AMD treatment sites within and immediately downstream of the mined area above Ravine, Pa., are used for this evaluation.

Second, was to test the hypothesis that AMD treatment (including land reclamation) has improved downstream water quality and promoted the return of fish and other aquatic life to the upper Swatara Creek and its major tributaries during the period 1996-2007. The paper evaluates a unique combination of data from annual surveys of fish populations; some benthic macroinvertebrate sampling; and continuous records of streamflow, temperature, pH, and other chemical data for stream segments downstream from AMD sources during the study period.

Study Area Description

Project Area

The Swatara Creek watershed is located in Schuylkill County, Pennsylvania (Figure 1). Swatara Creek drains an area of 1,472 km^2 in the Ridge and Valley Physiographic Province of eastern Pennsylvania, flowing 115 km from its headwaters in the Southern Anthracite Coalfield of Schuylkill County to its mouth on the Susquehanna River at Middletown, Dauphin County (Berg et al. 1989; McCarren et al. 1964). Approximately 75 % of the 112-km^2 (43 mi^2) area of the upper Swatara Creek Basin, upstream from the U.S. Geological Survey (USGS) streamflow-gaging station at Ravine (See Figure 2, USGS station 01571820), is underlain by anthracite-bearing bedrock. During the late 1800s through the 1940s, extensive underground mines were developed to depths as great as 1,000 m (Eggleston et al. 1999; Wood et al. 1968, 1986).

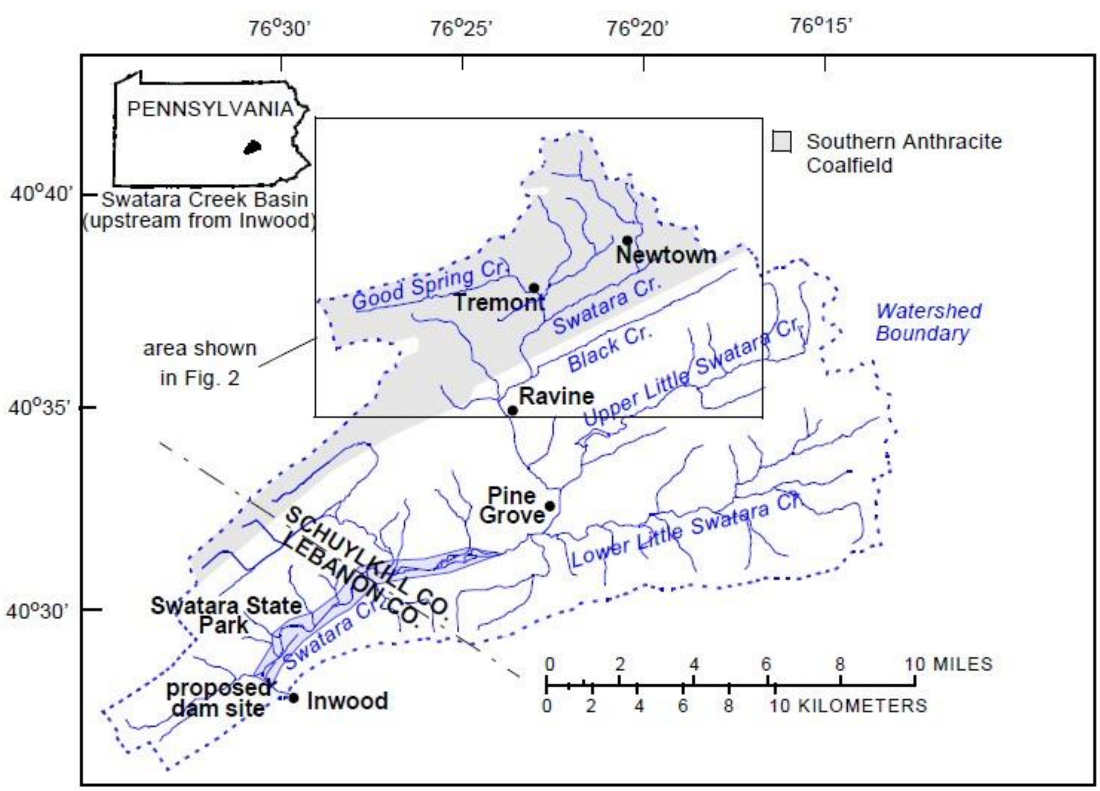

Figure 1: Location of northern Swatara Creek study area, upstream from the proposed dam for

Swatara State Park Reservoir, Lebanon and Schuylkill Counties, Pennsylvania.

Relevant Hydrologic, Geologic, and Meteorologic Factors

The northern Swatara Creek watershed drains the Southern Anthracite Field in the Ridge and Valley Physiographic Province. The watershed is underlain by siliciclastic bedrock of the Llewellen and Pottsville Groups. The ridges are held up by quartzite sandstone and conglomerate, whereas mostly softer rocks, including shale and siltstone with some interbeds of sandstone and anthracite, underlie the hillslopes and valleys. The mining of coal has had a significant effect on the watershed hydrology, affecting both the flow and quality of surface and ground water. Average annual rainfall for the watershed area is approximately 112 cm/yr (44 in/yr), with approximately 84 cm/yr (33 in/yr) of snowfall.

Land Use

Current land use in the upper 112-km^2 area is classified as 86.6 % forested, 4.9 % agricultural, and 6.4 % "barren, mined" (U.S. Geological Survey 2000). However, the land-use classification for this extensively mined area is misleading because underground mines extend beneath much of the surface and "natural" reforestation conceals large tracts of unreclaimed spoil. Agricultural development predominates downstream from the mined area. For example, land use in the area of the Swatara Creek Basin upstream from Pine Grove, is classified as 69.7 percent forested, 25.0 percent agricultural, and 2.4 percent barren, mined.

Water Resource Type and Size

The northern Swatara Creek watershed contains approximately 60 km (37 mi) of streams that will discharge to a proposed water-supply reservoir located in Swatara State Park. The proposed reservoir was planned to support recreational activities as well, including boating, fishing, and swimming. The water quality of source streams must be improved for the proposed reservoir to support all its designated uses.

Water Uses and Impairments

The streams of the northern Swatara Creek watershed are classified as cold-water streams. The Pennsylvania Fish and Boat Commission manage some of the streams as put-and-take trout waters. Additionally, the proposed reservoir to be constructed within Swatara State Park will support recreational activities including boating, fishing, and swimming.

AMD is considered to be the leading cause of degraded water quality in the project area. Acidity and high levels of sulfates and metals have created conditions that are toxic to some aquatic organisms. Recent efforts have been undertaken by the Pennsylvania Department of Environmental Protection (PaDEP), Bureau of Mining and Reclamation (BMR) to develop a watershed remediation plan. An EPA approved Watershed Implementation Plan has been written for the watershed which makes it a priority for Section 319 Nonpoint Source Funds. The goal of these plans is to improve water quality and restore the streams to recreational and fishable waters.

Pollutant Sources

AMD is the primary nonpoint source of pollution in the northern Swatara Creek basin; other sources are negligible. Although several surface and underground anthracite mines presently are active, most mines in the Swatara Creek Basin were abandoned before 1960. Barren, steep banks of spoil and culm and fine coal debris in siltation basins are sources of sediment (suspended solids), sulfate, iron, aluminum, and other metals in water that infiltrates or runs off the surface during storms. The abandoned underground mines have flooded and have collapsed locally causing subsidence. Surface flow is diverted through subsidence pits, fractures, and mine openings to the underground mines where the water becomes contaminated with acidity, sulfate, and metals. In downstream reaches, the contaminated water resurges as AMD contaminating Swatara Creek and its tributaries, while contributing substantially to baseflow.

A substantial proportion of the total streamflow originates as AMD. This source is most important during baseflow conditions. In contrast, during stormflow conditions, as much as 95 percent of the total streamflow for Swatara Creek at Ravine originates as surface runoff. The surface runoff typically has lower pH and lower concentrations of dissolved solids than the baseflow at Ravine.

Pre-Project Water Quality

Water quality data collected at 49 stations by BMR, Skelly and Loy Engineering Consultants, and the Northern Swatara Creek Watershed Association (NCSWA) volunteers from previous investigations were used to help document stream conditions and identify problem areas prior to installation of passive treatment systems. Data from these previous investigations included analysis of typical AMD; metals, major ions, acidity, and alkalinity.

The data indicated that a substantial proportion of the total streamflow originates as AMD. The investigations also revealed that the majority of the aluminum load to the stream originates from the eastern areas of the watershed upstream from Route 209 near Newtown (Figure 1) and the majority of the iron load originates from western areas of the watershed, including the Rowe Tunnel and Tracy Airshaft which are significant sources of water to Lorberry Creek and Good Spring Creek, respectively.

Description of Land Reclamation

Since 2000 there has been a significant amount of resources directed to abandoned mine reclamation and AMD remediation by the PA Department of Environmental Protection Bureau of Abandoned Mine Reclamation (BAMR). Over $7.3 million have been spent on thirteen reclamation projects addressing the most significant abanonded mine land (AML) features in the watershed. The positive impacts of many of the projects listed below are difficult to measure. Several projects reclaimed abandoned stripping pits that capture surface water and direct it to the abandoned underground mines. By reestablishing the natural contour of the land, the surface water is able to runoff and remain on the surface thus preventing the formation of AMD. The surface water will also serve to dilute the acid mine drainage in the streams.

The following BAMR projects addressed some of the most severe abandoned mines in the watershed.

North Donaldson I, OSM 54(3703)101.1 - This project is located in Frailey Township and involved backfilling several strip pits, some as deep as 70 feet, and a mine opening utilizing 434,697 cubic yards of on-site and off-site borrow material. This project also included the installation of a weir on the Colket water tunnel outflow to obtain flow data. A total of 39.9 acres were reclaimed at this site and 1,800 feet of highwall were eliminated. Final Cost: $521,953.24. Started 10/04/99,
Completed 07/30/00.

North Donaldson, OSM 54(3703)102.1 - This 40-acre project eliminated 5,300 feet of dangerous highwall, ranging in depth from 20 to 100 feet, and two mine openings. The work involved backfilling and grading 590,000 cubic yards of on-site material, revegetation of 50 acres, and installing two elliptical, reinforced concrete pipes to convey storm water runoff under SR 4009. Final Cost: $ 813,874.40. Started 07/18/01,
Completed 04/03/03.

North Donaldson III, OSM 54(3703)103.1 - The site is located 2,000 feet northeast of the village of Donaldson. This 85-acre project involved backfilling and grading of 1,347,810 cubic yards of on-site material, backfilling 2,500 feet of highwall ranging in depth from 50 to 100 feet, backfilling 5 mine openings, and constructing 3,500 feet of rock and PVC lined channel. Final Cost $ 1,206,921.21. Started 07/12/00, Completed 09/10/01.

Tremont North, OSM 54(3024)101.1 - This 66.3-acre project is located north of Tremont and involved the grading of 383,003 cubic yards of material to backfill 2,950 feet of abandoned strip pits and one mine opening. Final Cost $ 289,195.60. Started 02/12/01, Completed 07/06/01.

Tremont North III, OSM 54(3024)103.1 - This 18.7-acre project is located north of Tremont and involved the grading of a quarter million cubic yards of material to backfill 2,200 feet of abandoned strip pits, ranging in depth from 30 to 70 feet, with 1,000 feet classified as dangerous highwall. The final graded area was revegetated with grasses, legumes and tree seed. Final Cost $ 167,140.15. Started 04/27/03, Completed 12/08/03.

Robinson, OSM 54(4208)201.1 - This project involved the drilling of two ten-inch diameter boreholes, each 130 feet deep, to relieve the pressure of an aquifer contaminated by mine drainage, thus reducing or eliminating the flooding of the basements and the backyards of local residents. Drainage pipe was installed and the mine drainage was directed into a limestone rock-lined ditch, discharging into Good Spring Creek. Final Cost $ 39,120.00. Started 05/28/03, Completed 06/26/03.

Blackwood West, OSM 54(3648)101.1 - The Peach Mountain vein in the Llewellyn formation of the Pennsylvania period was mined in this area. This project is located in Reilly Township, Schuylkill County. Reclamation of the 75.3-acre site, which is also located on State Game Lands 229, involved the grading of 635,245 cubic yards to backfill 7,900 feet of dangerous highwall ranging from 20 to 80 feet deep. Two wetlands totaling 2.7 acres were constructed to mitigate for the 0.15 acres of wetland vegetation affected by the project. Final Cost $ 522,446.39. Started 06/27/00, Completed 05/17/01.

Red Mountain, OSM 54(3022)101.1 - Mining at this site was conducted by Philadelphia and Reading Coal and Iron Company, and operations ceased prior to 1969. The Diamond #14 and Diamond #14 1/2 veins were mined in this area. Reclamation of the 26.4-acre site, which is on State Game Lands 229, involved the grading of 62,230 cubic yards of on-site material to backfill 6,000 feet of dangerous highwall. Revegetation of the site consisted of warm season grass mixtures and brush windrows to provide wildlife food and cover. Final Cost $ 116,557.01. Started 05/23/00, Completed 11/01/00.

Newtown South I, OSM 54(3650)101.1 - This 31-acre project, located on State Game Lands 229, involved the grading of 139,207 cubic yards of material to backfill strip pits and mine openings. A total of 7,200 feet of dangerous highwall and four mine openings were eliminated. Revegetation of the site consisted of various warm season grass mixtures and legumes. Brush windrows were also constructed in order to provide additional habitat. Final Cost $ 124,404.23. Started 06/08/00, Completed 08/02/00.

Newtown South II, OSM 54(3649)101.1 - This project is located in Reilly and Tremont Townships, Schuylkill County, on State Game Lands 229. Reclamation of the 21.6-acre site involved the grading of approximately 110,786 cubic yards of material to eliminate 1,200 feet of dangerous highwall ranging in depth from 20 to 50 feet, and one mine opening. The project also involved the construction of 0.5 acres of wetlands. Final Cost $ 187,367.60. Started 07/12/00, Completed 10/03/00.

Middle Creek South, AMD 54(4214)201.1 - This 49-acre project involved the backfilling and grading of approximately 1,400 feet of abandoned dangerous highwalls, some as deep as 100 feet, with one million cubic yards of material. Natural channel design techniques were used to reconstruct 1100 feet of Middle Creek, which for the first time in over 30 years, now flows on the surface of the land instead of being lost into the abandoned underground mine workings. Two wetland areas have been constructed and a concrete box culvert was installed to convey Middle Creek beneath a township road. Final Cost $ 1,361,547.17. Started 03/06/00, Completed 09/26/03.

Indian Head Passive Treatment, AMD 54(3024)102.1 - This 8.1-acre project utilized 74,984 cubic yards of on-site material to backfill 2,950 feet of dangerous highwall. A passive treatment wetland system consisting of a 3.5-acre aerobic wetland and settling basin was constructed on site to treat acid mine discharges from the Marshfield Slope Outfall and the Marshfield No. 2 Outfall. The two discharges drain the Indian Head Mine Pool. The treated water is then conveyed by a grass swale to Coal Run. Final Cost $ 203,534.50. Started 05/16/00, Completed 01/12/01.

Newtown West, OSM 54(3652)101.1 - The Newtown West project is located near the village of Newtown in Reilly Township. This project reclaimed 48 acres of abandoned strip mines that included 5,800 feet of dangerous highwalls ranging in depth from 30 to 70 feet. The work involved the backfilling and grading of 373,000 cubic yards of on-site material, culvert installation, drainage ditch construction and the creation of a 0.6-acre wetland. The entire site was revegetated with a mixture of grasses, legumes and tree seed to control erosion and benefit wildlife. Final Cost $ 405,659.50. Started 01/05/04, Completed 10/28/04.

Description of Treatment Systems

During 1995-2001, various passive and semi-passive treatment systems were installed at selected locations to neutralize the AMD or the streamwater at downstream sites within the northern Swatara Creek Basin upstream from Ravine, Pa. (Figure 2). Where access and space were available, the treatment systems were located immediately below the AMD source (anoxic limestone drain, oxic limestone drain, aerobic wetland); otherwise, the systems were located within the downstream reach (limestone sand, open limestone channel, limestone diversion wells) of the affected stream (Table 1). The treatment systems were installed and maintained by the Schuylkill Conservation District and the Northern Swatara Creek Watershed Association. Technical and financial support for the design, construction, and monitoring of the treatment systems were provided by the Pennsylvania Department of Environmental Protection (PaDEP), U.S. Department of Energy (USDOE), and USGS.

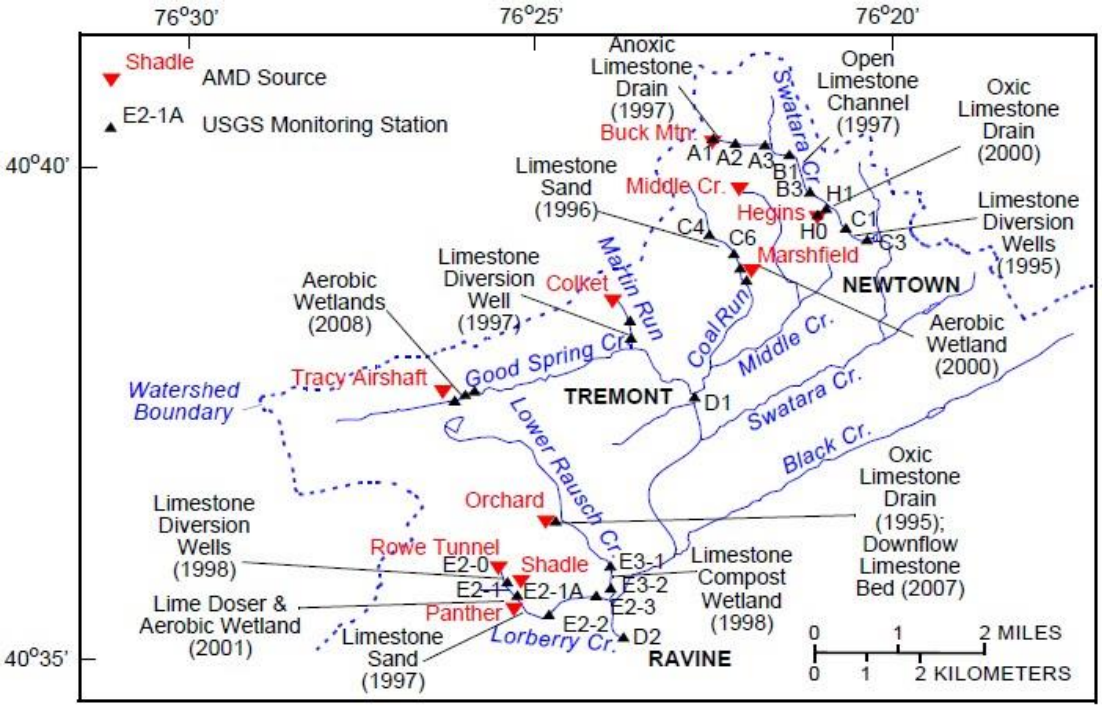

Figure 2: Locations of water-quality and streamflow monitoring sites relative to major AMD sources and associated treatment systems in the upper Swatara Creek Basin, above Ravine (See Fig. 1 for location of map area). Year of implementation of treatment is indicated in parentheses. Local monitoring site identification number (for example, E2-3A) is indicated for sites discussed in this paper. Official USGS station numbers for sites with continuous water-quality or streamflow data or fish data are as follows: C1 = 0157155010, Swatara Creek ab diversion wells; C3 = 0157155014, Swatara Creek bl diversion wells; D1 = 01571593, Good Spring Creek at Tremont; D2 = 01571820, Swatara Creek at Ravine; E2-2 = 01571778, Lorberry Creek at Mollystown; and E2-3 = 01571780, Lorberry Creek near Ravine.

Table 1. Acidic mine drainage (AMD) treatment system and associated water-quality monitoring sites, upper Swatara Creek Basin, Schuylkill County, Pennsylvania

USGS Station ID	Treatment System ID and Monitoring Site ID[1]		Latitude[2]	Longitude
Limestone sand in Coal Run below Middle Creek discharges (1996)				
0157158010	LSC	C4 (upstream)	403835	-762247
0157158014	LSC	C6 (downstream)	403832	-762246
Open limestone channel on Swatara Cr below Buck Mountain discharge (1997)				
0157154980	OLS	B1 (upstream)	404022	-762141
0157154984	OLS	B3 (downstream)	404022	-762136
Oxic limestone drain on Hegins discharge (2000, expanded 2005)				
403955076211801	ODH	H0 (influent)	403955	-762118
403955076211802	ODH	H1 (effluent)	403955	-762118
Anoxic limestone drain on Buck Mountain discharge (1997, expanded 2001 and 2005)				
404032076222901	ADB	A1 (influent)	404032	-762229
0157154970	ADB	A2 (effluent)	404032	-762225
0157154972	ADB	A3 (downstream)	404032	-762159
Limestone diversion wells on Swatara Creek below Hegins discharge (1995)				
0157155010	DWS	C1 (upstream)	403934	-762050
0157155014	DWS	C3 (downstream)	403928	-762043
Limestone diversion wells on Lorberry Creek below Rowe Tunnel discharge (1998)				
403542076263201	DWL	E2-0 (upstream)	403542	-762632
01571774	DWL	E2-1 (downstream)	403532	-762622
Aerobic wetlands beside Lorberry Creek below Rowe Tunnel discharge (2001)				
0157177610	WLL	E2-1A (influent)	403529	-762623
0157177618	WLL	E2-2 (effluent)	403527	-762619
Limestone-compost wetlands on Lower Rausch Cr below Orchard discharge (1998)				
01571758	WLR	E3-1 (upstream)	403522	-762442
01571760	WLR	E3-2 (downstream)	403534	-762440
Limestone sand in unnamed tributary to Lorberry Creek below Pantherhead discharge (1997)				
0157177780	n.a.[3]	n.a.	403510	-762556
Limestone diversion wells on Martin Run below Colket discharge (1997)				
0157156010	n.a.	n.a.	403819	-762419
0157156014	n.a.	n.a.	403816	-762419
Oxic limestone drain on Orchard discharge (1995, reconstructed as downflow limestone bed 2007)				
403626076253001	n.a.	n.a.	403626	-762530
403626076253026	ODO	n.a.	403626	-762529
Aerobic wetlands beside Coal Run below Marshfield discharges (2000)				
403828076224201	n.a.	n.a.	403828	-762242
0157158018	n.a.	n.a.	403823	-762246
Aerobic wetlands beside Good Spring Creek below Tracy Airshaft discharge (2008)				
0157177610	n.a.	n.a.	403529	-762623
0157177618	n.a.	n.a.	403527	-762619

a. AMD location, treatment system location, and monitoring site identification number are shown in Figure
b. Coordinates referenced to the North American Datum of 1983 (NAD 83). Values are degrees, minutes, seconds; 404032 represents 40°40'32" north latitude and -762229 represents 76°22'29" west longitude.
c. Treatment system and monitoring site identification numbers are indicated as "n.a." (not applicable) if the performance of the treatment system was not evaluated in this paper.

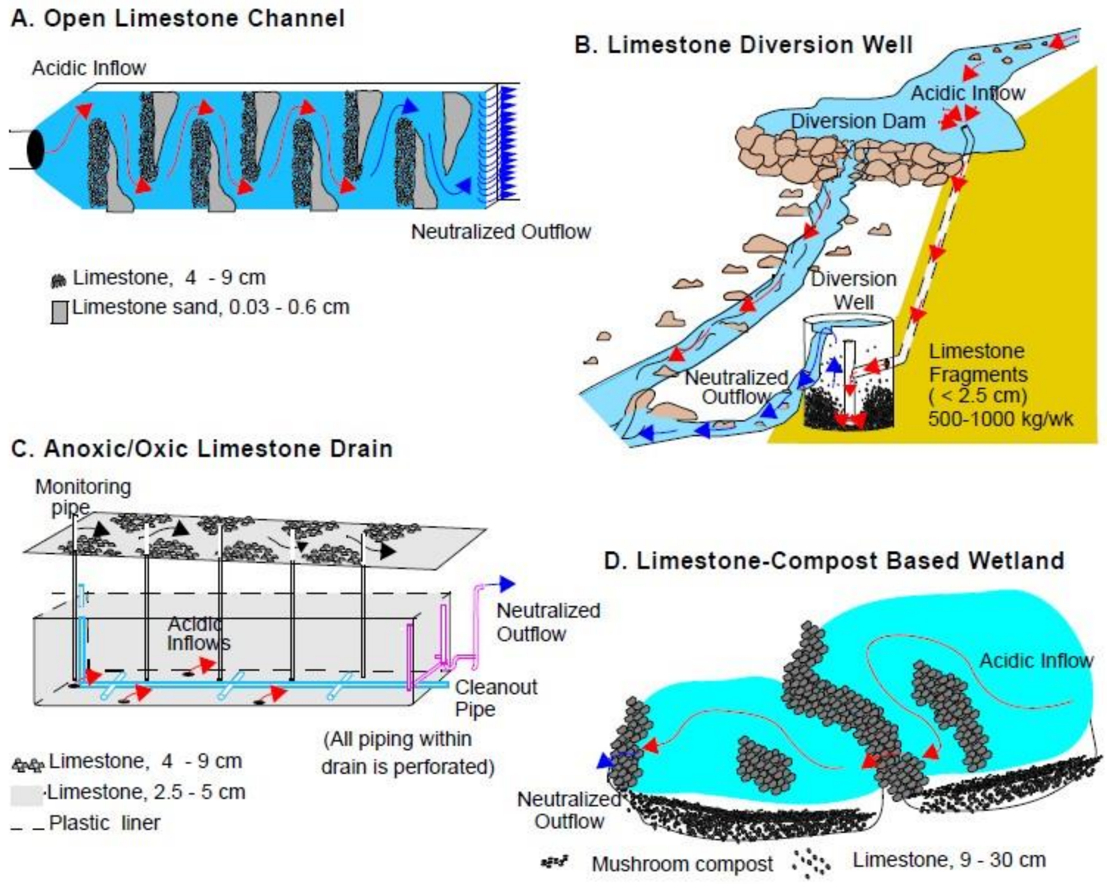

Figure 3 – Schematic illustrations of passive treatment systems installed in the upper Swatara Creek Basin, 1995-2001. Limestone fragment size designation (in parentheses) relates to aggregate size ranges of Pennsylvania Department of Environmental Protection (2000).

Limestone-sand dosing and open-limestone channels are relatively simple passive-treatment systems (Figure 3) where limestone is added once or infrequently to the streambed or AMD discharge channel (Skousen et al. 1998; Ziemkiewicz et al. 1997). Limestone sand, which can dissolve rapidly because of its small diameter, was dumped from trucks directly into Coal Run (40 metric tons) downstream from the Middle Creek discharges (Picture 3, Appendix A), between sites C4 and C6, in September 1996, and into an unnamed tributary of Lorberry Creek (136 metric tons) below the Panther discharge near site E2-1A in February 1997 (Figure 2). An open limestone channel was constructed within a 33.5 m long segment of Swatara Creek below the Buck Mountain discharge (Picture 1 and 2, Appendix A), between sites B1 and B3 (Figure 2), in March 1997. To construct the open limestone channel, a total of 40 metric tons of sand-size fragments (<0.5 cm) and 63 metric tons of larger fragments (3-11 cm) were installed as a series of alternating berms extending part way across the 4.6-m wide channel from opposite sides of the stream (Figure 3).

A limestone drain (Figure 3) is another relatively simple passive-treatment method that involves the burial of limestone aggregate in trenches that intercept acidic water (Cravotta and Trahan 1999; Cravotta et al. 2004; Hedin et al. 1994a, 1994b; Skousen et al. 1998). Keeping carbon dioxide (CO_2) within the limestone bed can enhance limestone dissolution and alkalinity production (Cravotta 2003; Cravotta and Trahan 1999). Keeping O_2 out of contact with the influent AMD minimizes the potential for oxidation of Fe^{2+} and the consequent precipitation of Fe^{III}-oxyhydroxide on the limestone surfaces or between particles. Although allowing O_2 into the limestone bed can facilitate the removal of Fe, Mn, and trace metals and accelerate limestone dissolution, the accumulation of Fe-rich solids can lead to clogging (Cravotta and Trahan 1999; Cravotta et al. 2004). Limestone drains designed for varying flow rates and chemistry were constructed in March 1995, at the Orchard discharge to treat a small oxic discharge (38-113 L/s; 40 metric tons limestone) along Lower Rausch Creek (Picture 4, Appendix A); in May 1997, at the Buck Mountain AMD (site A1) to treat a large, anoxic discharge (189-756 L/s; 320 metric tons limestone) at the headwaters of Swatara Creek (Picture 5 and 6, Appendix A); and in June 2000, at the Hegins AMD (site H0) to treat a large oxic discharge (378-1,890 L/s; 727 metric tons limestone) near the headwaters of Swatara Creek (Figure 2 and Picture 7 and 8, Appendix A).

Twice, in fall of 2002 and 2005, the anoxic limestone drain on the Buck Mountain AMD was enlarged with the addition of 91 metric tons of limestone. Additionally, in September of 2005, the oxic limestone drain on the Hegins discharge was enlarged with the addition of 182 metric tons of limestone and covered with approximately 0.15 m of leaf-litter compost. The enlargement and cover were intended to increase retention time, retain CO_2, and promote greater rates of limestone dissolution. Lastly, in August of 2007, the oxic limestone drain on the Orchard discharge, which had been out of service since 2000, was completely reconstructed as an upflow treatment system with flushing pipes and a settling basin to manage the accumulation of metal rich solids. The latter modifications were implemented after monitoring for the subject paper had been completed, so this system is not discussed hereinafter.

In a limestone diversion well (Figure 3), AMD is diverted from an upstream site into a pipe, and the hydraulic force at the terminus of the pipe is deflected upward through limestone aggregate inside 1.2-m diameter “wells” (Arnold 1991). As much as 1 metric ton of limestone can be consumed weekly by each operating diversion well, requiring regular replenishment of the limestone in this “semi-passive” system. Hydraulic churning within the diversion well abrades the limestone to fine particles and prevents encrustation by Fe(III) or Al oxyhydroxides. Dissolution of limestone within and downstream of the diversion well promotes increases in the pH and alkalinity of the stream. In addition to pulverized limestone, Fe(III) and Al oxyhydroxides may precipitate and accumulate downstream of the diversion wells. In November 1995, a pair of diversion wells was installed to treat water diverted from the headwaters of Swatara Creek below site C1 (Picture 9, Appendix A); in July 1997, a single diversion well was installed to treat water downstream from the Colket AMD on Martin Run below site C7; and in December 1998 a pair of diversion wells was installed to treat water downstream from the Rowe Tunnel AMD below site E2-0 near the headwaters of Lorberry Creek (Figure 4 and Picture 10, Appendix A). Because the Martin Run diversion well clogged repeatedly and was rarely working during the subject investigation, this system is not discussed hereinafter.

Constructed wetlands for treatment of AMD can attenuate the transport of dissolved and suspended pollutants by promoting the production of alkalinity and the precipitation and deposition of Fe and other metals (Cravotta 2007; Hedin et al. 1994a; Skousen et al. 1998). For net alkaline water, aerobic ponds and wetlands that facilitate the oxidation of Fe^{2+} and the settling of Fe(III) oxyhydroxides can be appropriate. For net acidic water, wetlands that have compost and/or limestone substrates can be useful to add alkalinity and remove dissolved metals. The organic matter in the compost provides a substrate for plant rooting and for microbial reduction of SO_4.

During 1997-2008, four wetlands were constructed to treat AMD in the upper Swatara Creek basin. In December, 1997, near the mouth of Lower Rausch Creek at site E3 (Figure 2 and Picture 11, Appendix A), a 0.93 ha compost-limestone based wetland was constructed to remove metals from streamflow that commonly had near-neutral pH but had potential to be net acidic during stormflow conditions (Koury and Hellier 1999). The Lower Rausch Creek wetlands were constructed downstream from the outflow of the Orchard oxic limestone drain built in 1995 (Figure 2, site E3). Although the Orchard oxic limestone drain was effective for neutralizing acid and converting dissolved metals to solid forms as described in another report (Cravotta and Trahan 1999; Cravotta et al. 2004), a settling basin or wetland was needed to attenuate the transport of suspended metals from the oxic limestone drain and other sites in the Lower Rausch Creek watershed. Additionally, in December 2001, 0.49-ha wetland was constructed near the confluence of Stumps Run and Lorberry Creek at station E2-1 (Figure 2 and Picture 10). The

Lorberry Creek wetland was constructed to remove iron from treated water exiting the two limestone diversion wells below the Rowe Tunnel discharge. Because the effluent from the Rowe Tunnel had widely variable pH, acidity, and metals concentrations, a hydrated lime doser was installed at the wetlands inflow to supplement the treatment by the diversion wells (Picture 12, Appendix A). Lastly, to treat the net-alkaline AMD from the Marshfield discharge along Coal Run and from the Tracy Airshaft discharge along Good Spring Creek, aerobic wetlands were constructed at these sites in June 2000 and May 2008, respectively. Because monitoring of the Marshfield and Tracy wetlands was not conducted as part of the subject investigation, this system is not discussed hereinafter.

Methods

Performance of Passive Treatment Systems

To characterize untreated AMD, treatment-system performance, and cumulative downstream effects of AMD treatment, the USGS established monitoring sites upstream and downstream of each treatment and along lower reaches of Swatara Creek (Figure 2). Fixed-interval grab samples (4-week or 6-week intervals) were collected over a range of hydrologic conditions from well-mixed zones at the stream and AMD monitoring sites. Instantaneous data on flow rate, temperature, specific conductance (SC), pH, redox potential (Eh), and dissolved oxygen (DO) were measured when water-quality samples were collected (e.g. Ficklin and Mosier 1999; Rantz et al. 1982a, 1982b). To minimize water-quality effects from aeration, AMD samples were collected and electrodes were immersed as close as possible to the point of discharge.

Whole-water subsamples were analyzed in the laboratory for alkalinity to pH 4.5 endpoint within 24 hours of sampling, whereas hot-peroxide acidity, total constituent concentrations, and "dissolved" (0.45-µm pore-size filter) subsamples were analyzed within 3 months of sampling (American Public Health Association 1998a, 1998b; Crock et al. 1999; Fishman and Friedman 1989; Hoffman et al. 1996). Because hot-peroxide acidity values obtained for this study did not include results for negative values, the computed net acidity (Kirby and Cravotta 2005), which counts positive contributions from H^+ and dissolved Fe, Mn, and Al and negative contributions from alkalinity, is evaluated in this paper. Data were stored in the USGS National Water Information System (NWIS) database (http://waterdata.usgs.gov/pa/nwis/qw), which is accessible to the public, and reported annually (U.S. Geological Survey, variously dated).

Hydrochemical data for influent and effluent samples or upstream and downstream samples for eight individual treatment systems were evaluated using graphical and statistical methods. If multiple samples were collected on a given date at a site, the daily average values were used. To provide temporal context for variable hydrologic conditions and seasonality, the upstream and downstream data for flow rate, pH, temperature, and concentrations of net acidity, dissolved Ca, dissolved SO_4, and dissolved and total Fe, Mn, and Al were illustrated as time-series plots. The overall effects of treatment were indicated by the differences between paired sample (downstream-upstream) data values for the different treatment systems during the post-implementation period. Boxplots were used to display the water-quality data by AMD site and the downstream-upstream differences for each treatment system.

The Wilcoxon matched-pairs signed-rank test (Helsel and Hirsch 2002) was used to indicate the significance of differences in water quality between upstream and downstream sites. The significance results of the signed-rank test were displayed as equality or inequality symbols above the boxplots showing the actual difference values between upstream and downstream data for each treatment. If the mean rank difference between the downstream site and the upstream site was insignificant at a probability level of 0.10, the difference would be equal to zero ("="). On the other hand, the treatment effects would be considered significant if the mean rank difference was positive (">") or negative ("<") at a probability level of 0.10. Furthermore, to indicate possible variability in treatment performance as a function of the hydrologic conditions, the rank differences also were evaluated for low-, normal-, and high-flow subsets. If streamflow of Swatara Creek at Ravine on the date of sampling was less than the 25th percentile for the study period, the sample was classified "low-flow"; between the 25th and 75th percentiles, the sample was classified "normal-flow"; or greater than the 75th percentile, the sample was classified "high-flow".

Data on treatment-system performance were normalized for comparison among different systems considering the acid-removal rate relative to the size and cost of the treatment system. In accordance with methods of Ziemkiewicz et al. (2003) for 83 different treatment systems in the eastern U.S., the acid-removal efficiency was computed as the median acid load removed (influent net-acidity load - effluent net-acidity load, in grams per day as $CaCO_3$) divided by the size of the treatment system. For wetland systems, the size was indicated by the total surface area (in square meters) of ponds and wetlands. For limestone systems, the size was indicated by the total mass of limestone (in metric tons) installed during the elapsed years in service. The mass of limestone for diversion wells was estimated as 30 metric tons per well for each year in service (each system had a pair of wells). In addition to acid load removed, the $CaCO_3$ load added was computed as 2.5 times the difference in dissolved calcium load from upstream to downstream. The cost efficiency was estimated to indicate the approximate cost per ton of acid treated over an assumed service life of 20 years. Because labor and materials for construction and maintenance of most of the treatment systems were donated or subsidized, the total cost for each treatment system was crudely estimated on the basis of the funds provided, equipment used, and the quantity of limestone and associated devices installed for treatment.

<u>Fish Data Collection and Analysis including Streamflow Monitoring</u>
To provide detailed information at a range of scales, the USGS collected hydrologic data at more than 80 locations in the upper Swatara Creek Basin during 1996-2007 (Szpir et al. 2007; U.S. Geological Survey, variously dated). For this paper, a subset of the monitoring data collected at primary streamflow-gaging stations on Swatara Creek, Good Spring Creek, and Lorberry Creek were utilized (Figure 2).

Fish were collected annually in Swatara Creek at Ravine and Newtown, Good Spring Creek at Tremont, and Lorberry Creek near Ravine (Figure 2) by electrofishing over a 150-m reach consisting of mixed riffle, run, and pool habitats (Barbour et al. 1999; U.S. Environmental Protection Agency 1993). Individual fish were identified and measured before releasing most specimens.

To evaluate the cumulative effects of AMD remediation and the transport of pollutants from the mined part of the upper Swatara Creek Basin to unmined areas downstream, in 1996, the USGS reestablished "continuous-record" stations for streamflow and water-quality monitoring on Swatara Creek at Ravine (USGS station 01571820; 1996-2007) near the outlet of the 112-km^2 upper basin, on Swatara Creek at Newtown (C3, USGS station 0157155014; 1996-2007) near the headwaters, and on Swatara Creek at Pine Grove (USGS station 01572025; 1996-2000) approximately 6 km downstream from the mined area. These sites had been monitored previously by USGS and others (Fishel 1988; Gannett Fleming Corddry and Carpenter, Inc. 1972; Growitz et al. 1985; McCarren et al. 1964; Skelly & Loy, Inc. 1987; Stuart et al. 1967). Additionally, continuous-record streamflow and water-quality gaging stations were established on Swatara Creek at Newtown (C1, USGS station 0157155010; 1996-2007), upstream of limestone diversion wells, and on Lorberry Creek at Mollystown (USGS station 01571778; 1999-2007) (Figure 2).

The continuous-record stations were equipped with automatic stage-recording, water-quality monitoring, and/or water-sampling devices. The stream stage was measured continuously with a pressure transducer, and the temperature, pH, and specific conductance (SC) were measured continuously with a multiparameter sonde (e.g. Wagner et al. 2000). The stage and water-quality values were recorded at 15- minute intervals. To estimate continuous streamflow, stage-discharge ratings were developed for each site on the basis of instantaneous streamflow for a range of stream stages (e.g. Rantz et al. 1982a, 1982b). Streamflow typically was measured by wading across the channel with a vertical-axis current meter.

Instantaneous data for temperature, SC, pH, redox potential (Eh), and dissolved oxygen (DO) were measured using standard field methods (e.g. Ficklin and Mosier 1999) when continuous-record data were retrieved at streamflow-gaging stations or when water-quality samples were collected. Fixed-interval grab samples, mostly at base-flow conditions, were collected at 4-week or 6-week intervals from well-mixed zones in the stream. For Swatara Creek at Ravine (USGS station 01571820), Swatara Creek at Newtown (USGS station 0157155014), and Lorberry Creek at Mollystown (USGS station 01571778), numerous additional base-flow and stormflow samples were collected using pumping samplers containing 24 1-L polyethylene bottles. Stormflow samples submitted for analysis were selected to cover rising, peak, and falling stages of the hydrograph. Stormflow samples of Swatara Creek at Ravine were analyzed for more than 60 events during the study. Bulk precipitation samples for a few of the storms also were collected and analyzed.

Water samples were split into subsamples in the field or in the USGS Pennsylvania Water Science Center laboratory and stored in sample-rinsed polyethylene bottles at 4°C. Whole-water samples were analyzed in the laboratory within 24 hours of collection for pH and "acid-neutralizing capacity" (alkalinity) to pH 4.5 endpoint (American Public Health Association 1998a). Samples for "dissolved" (filtered through membrane with 0.45-μm pore size) and total recoverable (whole-water; in-bottle digestion with nitric acid (HNO_3) and hydrochloric acid (HCl)) metal analysis were stored in acid-rinsed polyethylene bottles and acidified with HNO_3. The water samples were analyzed for major ions and trace metals by inductively coupled plasma atomic emission spectrometry (ICP-AES), ion chromatography (IC), colorimetry, and electrometric titration (Crock et al. 1999; Fishman and Friedman 1989; Hoffman et al. 1996) at the Pennsylvania Department of Environmental Protection (PaDEP) Bureau of Laboratories facility in Harrisburg, Pa., during 1996-2000, at the US Department of Energy laboratory in Pittsburgh, Pa., during 2001-2002, and at the Actlabs laboratory in Toronto, Ontario, during 2003-2007. Although similar analytical procedures were used, the laboratories reported different limits of detection for aluminum and trace metals. For quality assurance of chemical analyses, USGS standard reference water samples (SRWS) were submitted with each batch of samples. Data for environmental water-quality and SRWS samples collected during the study were stored in the USGS National Water Information System (NWIS) database (http://waterdata.usgs.gov/pa/nwis/qw), which is accessible to the public, and reported annually (U.S. Geological Survey, variously dated).

Hardness, expressed in milligrams per liter as $CaCO_3$, was computed from the concentrations of dissolved calcium and magnesium in milligrams per liter ($2.5 \cdot C_{Ca} + 4.1 \cdot C_{Mg}$). The net acidity, which is similar in value to the "hot peroxide" acidity (American Public Health Association 1998b), was computed considering positive acidity contributions from protons and concentrations of dissolved iron, manganese, and aluminum, and negative contributions from alkalinity as described by Kirby and Cravotta (2005). Because the hot acidity values obtained for this study did not include results for negative values, only the net acidity is evaluated in this paper.

Streamflow and water-quality data were evaluated using various graphical and computational methods to indicate frequency distributions, correlations, and trends. To compare hydrologic conditions among sites during the study with the long-term record, streamflow duration records (probability plots) for the Ravine and Newtown streamflow-gaging stations were displayed with records for stations on Swatara Creek at Pine Grove (USGS station 01572025) and Harper Tavern (USGS station 01573000), which were 7.7 and 48.0 km downstream from Ravine, respectively. Daily mean streamflow values for these sites also were used with the PART hydrograph-analysis computer program (Rutledge 1998) to estimate annual mean streamflow and base-flow and surface-runoff contributions during the study. Interbasin variability during the study was indicated by the streamflow "yield," computed by dividing the annual streamflow by the estimated drainage area at the gaging station. In units of centimeters per year, the streamflow yield can be compared with annual rainfall and used to indicate evapotranspiration (rainfall minus streamflow yield), recharge (base-flow yield), and other water-budget terms for the basin (Cravotta and Nantz 2008; Rutledge 1998). Hydrographs and time-series displays of water-quality data, such as boxplots and probability plots by time interval, were used to illustrate potential trends during the study. For graphical illustration, the instantaneous load (transport) was computed as the product of concentration and flow rate.

A multivariate approach was used to compute daily concentration and unbiased estimates of annual load at continuously gaged monitoring sites. This approach described by Langland et al. (2006) uses the log-linear 7-parameter “ESTIMATOR” regression model of Cohn et al. (1989) with daily mean streamflow and time parameters to estimate the continuous distribution of daily concentration values:

$$\ln(C) = ß_0 + ß_1\ln(q/q_c) + ß_2[\ln(q/q_c)]^2 + ß_3(t-t_c) + ß_4(t-t_c)^2 + ß_5\sin(2\pi t) + ß_6\cos(2\pi t) + \varepsilon \quad (1)$$

where

ln is the natural logarithm function;
C is measured concentration, in milligrams per liter;
q is measured daily mean streamflow;
t is time, in decimal years;
q_c, t_c are centering variables for streamflow and time;
$ß_i$ are coefficients estimated by ordinary least squares (non-censored observations) and Adjusted Maximum Likelihood Estimator (Cohn 1988) (censored observations);
$ß_0$ is a constant;
$ß_1$, $ß_2$ describe the relation between concentration and streamflow;
$ß_3$, $ß_4$ describe the relation between concentration and time, independent of streamflow;
$ß_5$, $ß_6$ describe seasonal variation in concentration data; and
ε is residual error, assumed to be normally distributed with a standardized variance of one.

After determining the regression coefficients on the basis of measured concentrations and streamflow, the daily mean streamflow values were used with Eq. (1) to estimate daily concentrations of hydrogen ion (pH), alkalinity, dissolved calcium, dissolved sulfate, dissolved and total iron, dissolved and total manganese, dissolved aluminum, and dissolved zinc for Swatara Creek at Ravine (01571820) and Swatara Creek at Newtown (0157155010 and 0157155014). Daily streamflow data for other sites were not available for the 10-year record considered necessary for this method to be useful (M. J. Langland, U.S. Geological Survey, 2008, oral commun.). The daily concentration estimates for the 1997-2006 period were then multiplied by daily mean streamflow and integrated over time to indicate annual loads. Next, by dividing the annual load by the annual streamflow, the annual mean flow-weighted concentration was computed for each calendar year of the study. The flow-weighted concentration computed on this basis is considered an unbiased estimate of the mean concentration in a total volume of water flowing past a specific location in a specific time period (Langland et al. 2006).

Flow-adjusted trends were estimated by considering only the coefficients ($ß_3$, $ß_4$) for the time terms in the log-linear regression model (Eq. 1). As explained in detail by Langland et al. (2006), the flow-adjusted trends, expressed as percent difference relative to the starting time, indicate the overall change between the start date (1997) and the end date (2006) and are mathematically identical for concentration and load. Changes were considered significant only if the confidence interval of the modeled value at the end of 2006 was entirely greater than (upward trend) or entirely less than (downward trend) the modeled starting value. The results of flow-adjusted trend analysis can be interpreted to indicate changes in water quality that result from factors other than streamflow, such as changes in land use or other management practice (Helsel and Hirsch 2002).

Macroinvertebrate Collection

Macroinvertebrates were collected on Swatara Creek at Ravine during the fall of 1994 and 1996-2000. In accordance with rapid bioassessment protocols (U.S. Enironmental Proetection Agency 1990; Barbour *et al.* 1999), a rectangular frame kicknet with 0.6-mm screen size was used to capture debris and organisms dislodged from the streambed. An area of approximately 0.5 m^2 was ‘kicked’ upstream of the net for a total of 30 seconds for each sample. Samples were collected from three habitats consisting of shallow riffle with exposed cobbles, deeper riffle and run habitats. These three samples were composited and preserved with a formaldehyde solution for subsequent identification in the laboratory.

Results

Characterization of AMD Sources and Effects on Streamwater Quality

Although more than 40 AMD sources in the upper Swatara Creek Basin had been identified during previous investigations, most were minor sources of contaminant loading (Growitz et al. 1985; Skelly & Loy, Inc. 1987). The major AMD sources studied during the previous and current investigations had large flow rates (medians greater than 100 L/min), such as the Tracy Airshaft, Rowe Tunnel, Middle Creek, Colket, Buck Mountain, and Hegins discharges, and/or elevated concentrations of dissolved metals, such as the Pantherhead, Shadle, and Orchard discharges (Fig. 4, Table 1). Generally, the AMD sources with the largest flow rates during the 1996-2007 study had slightly acidic or near-neutral pH (>5) and elevated concentrations of dissolved Fe (> 3 mg/L). Depending on the AMD source, the flow rate at a given site varied by 1 to 3 log units during the 1996-2007 study; associated chemical variations were less pronounced. The larger volume AMD sources generally had the least-variable flow rates and chemistry, with slightly acidic or near-neutral pH (> 5) and elevated concentrations of dissolved Fe (> 3 mg/L). In contrast, the smaller volume AMD sources had the most variable flow rates and chemistry, with moderately acidic to strongly acidic pH (< 4.5) and elevated concentrations of Al, Ni, and Zn (0.1 to 1 mg/L) (Table 2). Concentrations of Mn typically were greater than or equal to 1 mg/L for all the AMD sources. Elevated concentrations of dissolved Mn and Fe, independent of pH (Figure 4), generally indicate redox-controlled, kinetic limitations on the precipitation of oxidized compounds of these metals, whereas decreased concentrations of dissolved Al with increased pH and decreased concentrations of dissolved Zn with increased pH are consistent with solubility control by Al-hydroxide and sorption control by Fe^{III} oxyhydroxide, respectively (e.g. Cravotta 2008a).

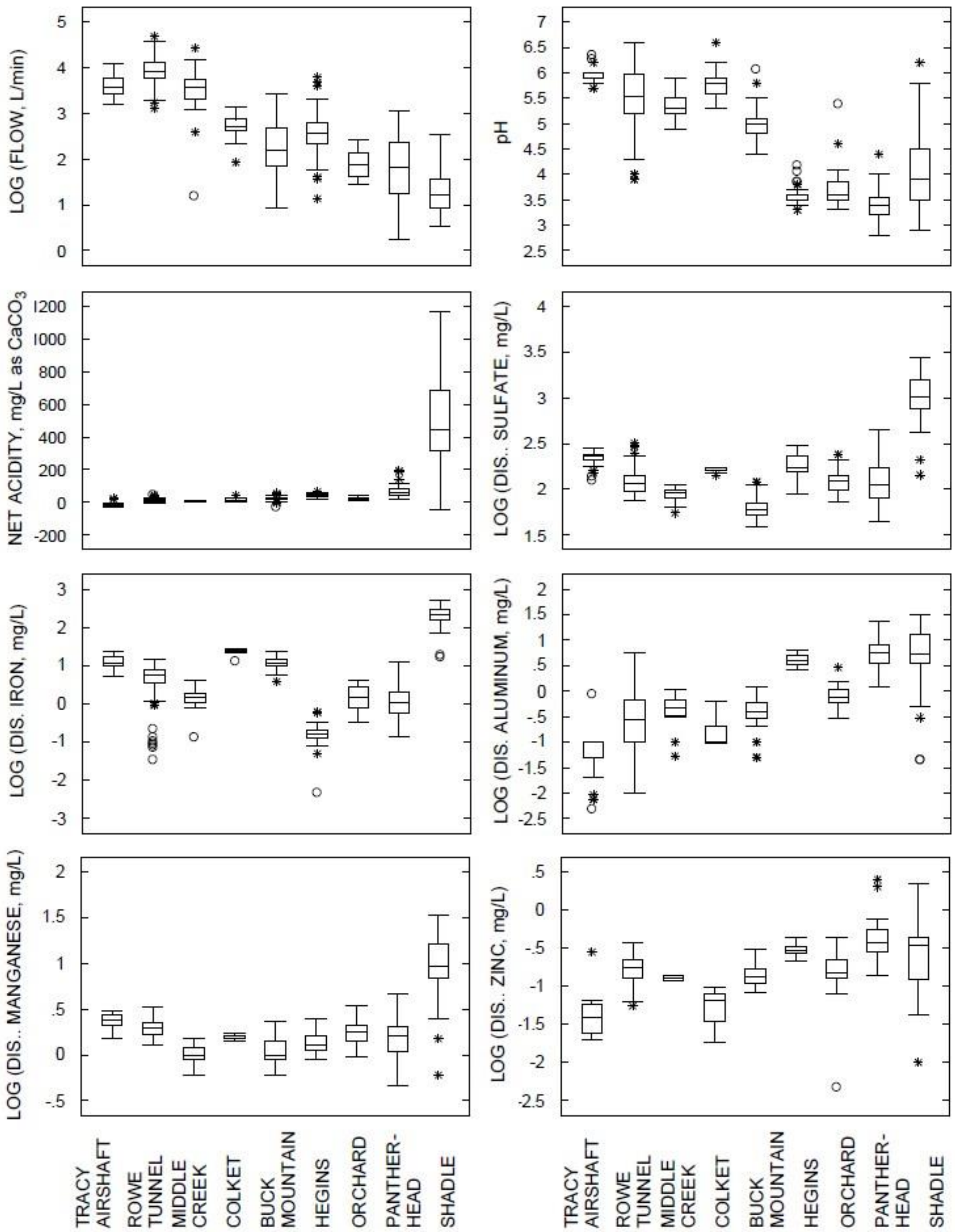

Figure 4. Boxplots summarizing hydrochemical characteristics of AMD sources upstream from any treatment in the Swatara Creek Basin, Pa., 1996-2007. Area of box indicates the "interquartile" range (IQR = 25th to 75th percentile); horizontal line inside the box indicates the median; vertical lines extend to extreme values within 1.5 times the IQR; symbols indicate outlier values.

Table 2. Median water quality and constituent loading for AMD in upper Swatara Creek Basin, 1996-2007 [L/min, liters per minute; ºC, degrees Celsius; µS/cm, microsiemens per centimeter; mg/L, milligrams per liter; kg/d, kilograms per day; dis., dissolved (<0.45 µm); <, less than]

	AMD Sites									
Constituent	Tracy Airshaft	Rowe Tunnel	Middle Creek	Marsh-`field[4]	Colket	Buck Mountain	Hegins	Orchard	Panther-head	Shadle
Number of observations	31	134	19	1	17	45	54	20	60	72
Flow rate (L/min)	3740	8310	3740	1400	501	132	374	77	51	17
Temperature (°C)	11	12	11	9.32	11.5	10.2	10	10.8	9.7	12.8
SC (µS/cm)	91	301	256	361	419	206	431	333	343	1730
DO (mg/L)	1.0	9.4	8.4	3.6	5.4	1.4	10.3	5.8	8.4	1.0
pH (units)	5.9	5.5	5.3	6.4	5.8	5.0	3.5	3.6	3.4	3.9
Net acidity (mg $CaCO_3$/L)	-15.5	9.3	4.6	-59	12.5	21.9	38.7	21.5	57.5	443
Alkalinity (mg $CaCO_3$/L)	43	4	3	74	30	4	0	0	0	0
SO_4, dis. (mg/L)	230	115	92.3	120	160	60.6	174	123	110	1030
Ca, dis. (mg/L)	40	13	12.3	37	29	3.8	7.8	16.8	8.3	160
Fe, dis. (mg/L)	12	5.89	1.5	6.4	23	11.5	0.16	1.46	1.11	219
Al, dis. (mg/L)	<0.10	0.28	0.467	<0.10	<0.10	0.40	4.0	0.76	5.7	5.2
Mn, dis. (mg/L)	2.4	2.0	1.0	1.7	1.6	1.0	1.3	1.8	1.6	9.3
Ni, dis. (mg/L)	0.055	0.075	0.053	0.025	0.061	0.070	0.111	0.084	0.140	0.122
Zn, dis. (mg/L)	0.038	0.175	0.130	0.012	0.064	0.132	0.295	0.150	0.375	0.344
Net acidity (kg $CaCO_3$/d)	-91	110	24	-103	10	4	22	2	6	9
SO_4, dis. (kg/d)	1120	1550	470	222	121	12	90	12	11	15
Ca, dis. (kg/d)	217.0	166.0	69.9	62.3	23.3	1.0	4.1	1.6	0.8	3.8
Fe, dis. (kg/d)	88.1	65.5	9.4	7.2	19.2	2.4	0.1	0.2	0.1	4.9
Al, dis. (kg/d)	0.44	4.06	2.42	0.20	0.12	0.09	2.04	0.06	0.59	0.10
Mn, dis. (kg/d)	13.2	24.0	5.3	2.9	1.2	0.2	0.7	0.2	0.1	0.2
Ni, dis. (kg/d)	0.281	0.965	0.227	0.050	0.052	0.014	0.065	0.006	0.015	0.002
Zn, dis. (kg/d)	0.160	2.23	0.599	0.020	0.025	0.028	0.175	0.014	0.039	0.005

a. Only one water-quality sample with flow data was available for Marshfield Discharge before a wetland constructed in 2000 flooded the site, preventing access.

During the 1996-2007 study period, the Shadle discharge exhibited the widest variability in pH and chemical concentrations compared to other AMD sources in the watershed (Figure 4). The pH of the Shadle discharge increased progressively from values of 3.1 to 3.2 in 1996-1998 to values of 4.9 to 6.2 in 2005- 2007; corresponding concentrations and loadings of net acidity, SO_4, Al, Fe, and other contaminants decreased. Although a decrease in contaminant loads from an AMD source would be anticipated with treatment, the improved quality of the Shadle discharge over the study period did not result from treatment but resulted from the rapid flooding of this underground mine following its closure around 1990. Permanent flooding of a mine can result in the (1) dissolution of accumulated pyrite oxidation products, (2) reduction in the access of oxygen to the subsurface with a corresponding decrease in the pyrite oxidation rate, and (3) progressive dilution of initially acidic water, potentially by alkaline groundwater. Extensive flooding of underground mines throughout the region and the gradual balancing of acidity and alkalinity can account for "natural" improvement in AMD and surface-water quality that has been ongoing for decades, particularly in the Northern, Western, and Southern Anthracite Coalfields (e.g. Raymond and Oh 2009; Wood 1996).

Despite evidence for natural attenuation of AMD contamination in the Swatara Creek Basin, downstream conditions generally were marginal for aquatic biota prior to the implementation of treatment systems. During 1996-2000, streamwater of Swatara Creek at Newtown and Ravine (Figure 1 and 2) ranged from mildly acidic to near-neutral (pH 4.5 to 8.0) with moderate concentrations of dissolved solids (SC 60 to 400 μS/cm) that varied as a function of streamflow (Cravotta and Weitzel 2001; Cravotta and Bilger 2001). Higher values of pH, SC, and SO_4 were associated with base-flow conditions sustained by near-neutral groundwater and AMD in the upper part of the watershed, such as the Tracy Airshaft and Colket discharges (Figure 4). Lower values of pH, SC, and SO_4 were associated with increased flows from acidic AMD sources such as the Buck Mountain, Hegins, and Pantherhead discharges plus mixing of base flow with acidic storm runoff (Cravotta 2000).

In contrast with Swatara Creek at Newtown and Ravine, the pH and SC for headwaters of Lorberry Creek, below the Rowe Drainage Tunnel (Figure 1 and 2), were inversely correlated with each other and varied widely (Cravotta and Weitzel 2001). Instead of storm runoff as the primary cause of variations, periodically pumped AMD with low pH and elevated concentrations of SO_4 and other dissolved ions caused increased flows, decreased pH, and increased SC of Lorberry Creek. Although the Rowe Drainage Tunnel drained an abandoned mine complex, an underground mine that was active below the complex during the study regularly pumped untreated, acidic water to the overlying mine pool. On the shorter scale of a few days, regular fluctuations in water quality resulted from the addition of this untreated AMD. When the pumping was active, pH of Lorberry Creek declined by 0.5 to 1 unit, and SC increased by 50 to 200 μS/cm. These short-term fluctuations in pH and SC were apparent for the continuous monitoring data on Lorberry Creek at Mollystown and for Swatara Creek at Ravine, particularly during base-flow conditions (Cravotta and Weitzel 2001). Because multiple AMD sources and acidic storm runoff were possible causes of impairment, treatment systems were implemented along stream reaches downstream from the AMD and at specific AMD sources, where access and space were not limiting (Figure 2, Table 1).

<u>Evaluation of Treatment Performance</u>

Generally, all eight of the treatment systems that were evaluated removed acidity, as indicated by the negative difference (downstream - upstream) in net acidity load (Figure 5). However, the acid load treated (Tables 3 and 4) and the magnitude of effects, if any, on the flow rate, pH, temperature, and loading of dissolved Ca, dissolved SO_4, and dissolved and total Fe, Mn, and Al varied widely among the treatment systems (Figure 5).

Table 3. Median influent and effluent quality, acid load removed, costs, removal efficiency (R.E.), and cost efficiency (C.E.) for limestone-based treatment systems in upper Swatara Creek Basin

[L/s, liters per second; mg/L, milligrams $CaCO_3$ per liter; t/y, metric tons $CaCO_3$ per year; g/d/t, grams acidity as $CaCO_3$ removed per day per ton of limestone; $/t, dollars per ton of acidity as $CaCO_3$ removed]

Treatment[5]	Number of sample pairs	Flow (L/s)	pH		Net Acidity (mg/L)		Acid Load Removed (t/y)	$CaCO_3$ Load Added (t/y)	Size	Years in Service	Cost[6] ($)	R.E. (g/d/t)[7]	C.E. ($/t)[8]
			Influent	Effluent	Influent	Effluent			Mass (t)				
LSC	11	56.6	6.5	6.5	-8.5	-8.8	-1.9	2.2	40	11	1,500	47.5	108
OLS	39	62.3	6.4	6.5	-2.8	-3.0	-0.2	0.1	100	10	3,500	2.0	2,397
ODH	50	5.4	3.5	4.6	38.2	23.2	-7.7	3.0	909	7	50,000	8.5	891
ADB	38	24.6	5.0	6.6	23.2	-10.1	-26.2	27.9	502	10	25,000	52.1	131
DWS	82	82.4	5.5	6.0	0.0	-1.6	-9.1	8.6	840	12	68,800	12.7	1,035
DWL	64	131.7	5.5	6.2	9.3	6.2	-40.9	1.3	630	9	68,800	75.8	230

a. Treatments described in Table 1: LSC, limestone sand in Coal Run; OLS, open limestone channel on Swatara Creek; ODH, oxic limestone drain at Hegins discharge; ADB, anoxic limestone drain at Buck Mountain discharge; DWS, limestone diversion wells on Swatara Creek; DWL, limestone diversion wells on Lorberry Creek.

b. The cost to build and maintain the diversion wells was estimated assuming $20,000 for initial installation of the wells with hopper storage, $24 per metric ton for limestone over 20 years, and $1,000 annually for operation and maintenance (filling wells, clearing debris from intakes).

c. R.E. computed as acid load removed divided by mass of limestone, multiplied by conversion factor of 1,000 g/kg.

d. C.E. computed as total cost divided by acid load removed assuming 20 year service life, multiplied by conversion factor of 2.74 (d·t)/(y·kg).

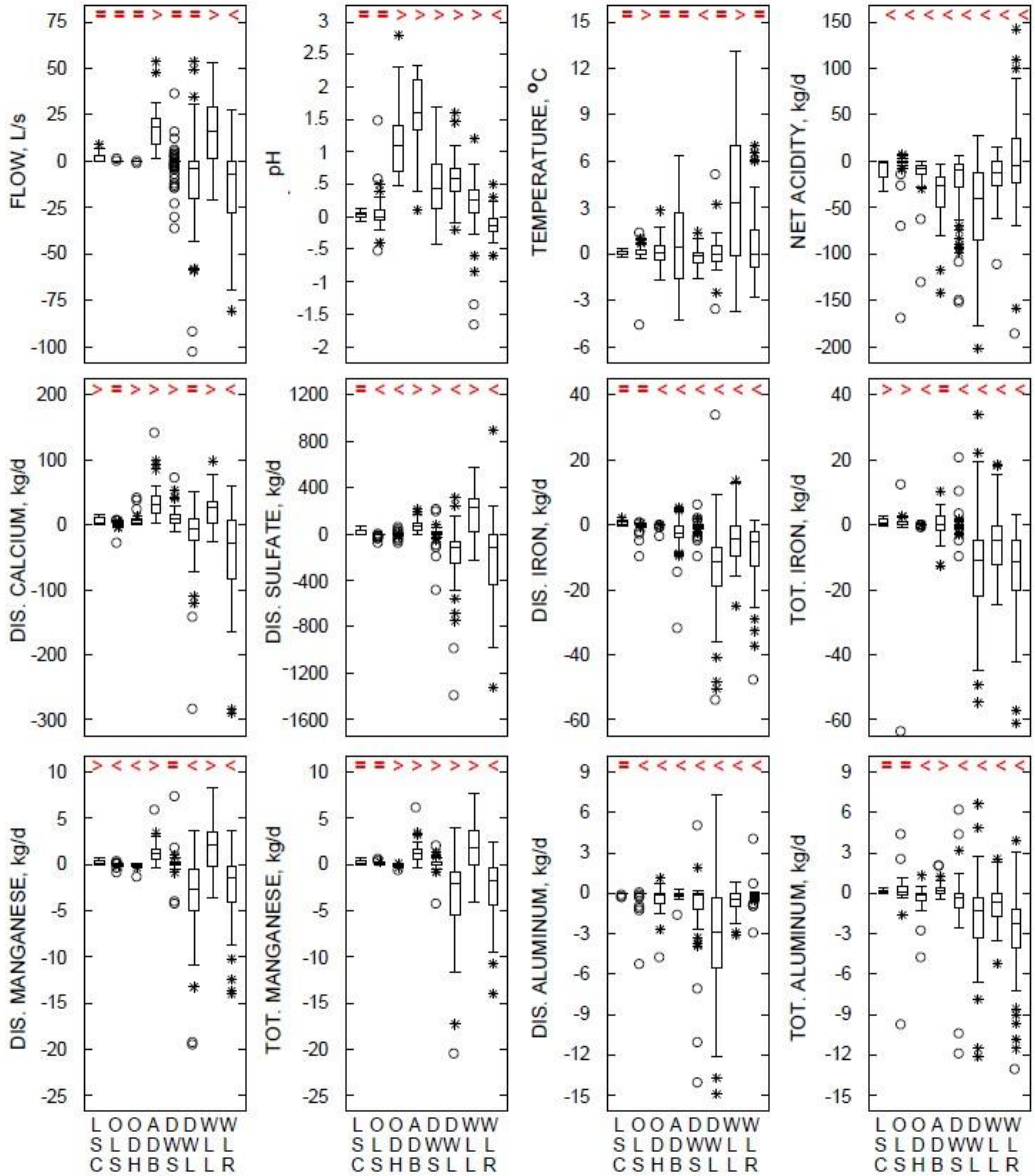

Figure 5. Boxplots of post-implementation data showing difference in water quality between downstream and upstream sites for selected treatment systems in Swatara Creek Basin. Axis labels are: LSC, limestone sand in Coal Run (C6 - C4); OLS, open limestone channel on Swatara Creek (B3 - B1); ODH, oxic limestone drain at Hegins discharge (H1 - H0); ADB, anoxic limestone drain at Buck Mountain discharge (A3 - A1); DWS, limestone diversion wells on Swatara Creek (C3 - C1); DWL, limestone diversion wells on Lorberry Creek (E2-1 - E2-0); WLL, aerobic wetlands on Lorberry Creek (E2-2 - E2-1); WLR, aerobic limestone-compost wetlands on Lower Rausch Creek (E3-2 - E3-1). See explanation of boxplot symbols in Figure 3. Symbols at the top of each boxplot indicate if the mean rank at downstream site was equal to (=), greater than (>), or less than (<) that for the upstream site on the basis of the Wilcoxon matched-pair signed-rank test (Helsel and Hirsch 2002).

Table 4. Median influent and effluent characteristics, acid load removed, costs, removal efficiency (R.E.), and cost efficiency (C.E.) for wetland treatment systems in upper Swatara Creek Basin
[L/s, liters per second; mg/L, milligrams $CaCO_3$ per liter; t/y, metric tons $CaCO_3$ per year; m^2, square meters; $g/d/m^2$, grams acidity as $CaCO_3$ removed per day per square meter; $/t, dollars per ton of acidity as $CaCO_3$ removed]

Treatment[9]	Number of sample pairs	Flow (L/s)	pH		Net Acidity (mg/L)		Acid Load Removed (t/y)	$CaCO_3$ Load Added (t/y)	Size	Years in Service	Cost ($)	R.E. ($g/d/m^2$)[10]	C.E. ($/t)[11]
			Influent	Effluent	Influent	Effluent			Area (m^2)				
WLL	33	56.1	5.7	6.0	6.5	0.6	-12.3	23.8	4,860	6	142,000	2.5	1,584
WLR	61	157.5	7.0	6.8	-11.5	-12.9	-3.1	-27.0	9,310	9	175,000	0.3	7,783

a Treatments described in Table 1: WLL, aerobic wetlands on Lorberry Creek; WLR, limestone-compost wetlands on Lower Rausch Creek.

b R.E. computed as acid load removed divided by area of wetland, multiplied by conversion factor of 1,000 g/kg.

c C.E. computed as total cost divided by acid load removed assuming 20 year service life, multiplied by conversion factor of 2.74 (d·t)/(y·kg).

For example, the limestone drains at the Hegins and Buck Mountain discharges (ODH, ADB) and limestone diversion wells on Swatara Creek and Lorberry Creek (DWS, DWL) increased the pH and decreased the dissolved Fe and Al load downstream; however, the limestone sand on Coal Run (LSC) and the open limestone channel on Swatara Creek (OLS) had only minor effects, if any, on the pH and dissolved metals loads. Likewise, the two wetland systems along Lorberry Creek and Lower Rausch Creek (WLL, WLR) decreased dissolved Fe and Al loads, but had varying effects on pH. The results of treatment by individual systems and factors affecting their performance are described below.

Limestone-Sand Dosing on Coal Run (LSC; C4 - C6). The limestone-sand dosing at Coal Run was aptly called dumping (Picture 3, Appendix A), whereby several truckloads of finely crushed limestone were spilled at once over the streambank into the channel. As the mound of limestone sand was eroded at the base, fresh limestone spilled into the channel where it gradually dissolved. Streamflow in the treated section of Coal Run, which originated as AMD from several sources similar in quality to the Middle Creek and Marshfield discharges (Figure 4, Table 2), ranged from 8.5 to 215 L/s. After implementation of treatment, the untreated streamwater had pH values from 5.6 to 6.9, net acidity concentrations from -11.2 to 1.4 mg/L $CaCO_3$, and moderate concentrations of dissolved metals (Fe 0.5 to 2.0 mg/L; Mn 0.76 to 1.2 mg/L; Al < 0.5 mg/L) (Figure 6). The magnitude of water-quality differences between the upstream site (C4) and downstream site (C6) was small (Figures 6 and 7). Although the pH and dissolved concentrations and loads of Fe and Al were not significantly different between the upstream and downstream sites for most conditions, the matched-pair tests (Figure 5) indicated the net acidity decreased and dissolved Ca and Mn concentrations increased downstream. (Mn could be an impurity in the limestone.) The limestone-sand treatment on Coal Run removed a median acidity of 1.9 t/y and added 2.2 t/y Ca as $CaCO_3$ and continued to provide benefits over the 6-year monitoring period (Figure 6, Table 3). Compared to the other seven treatment systems, the limestone-sand treatment of Coal Run had relatively high acid-removal efficiency (47.5 g/d/t) and the best cost efficiency ($108/t) (Table 3).

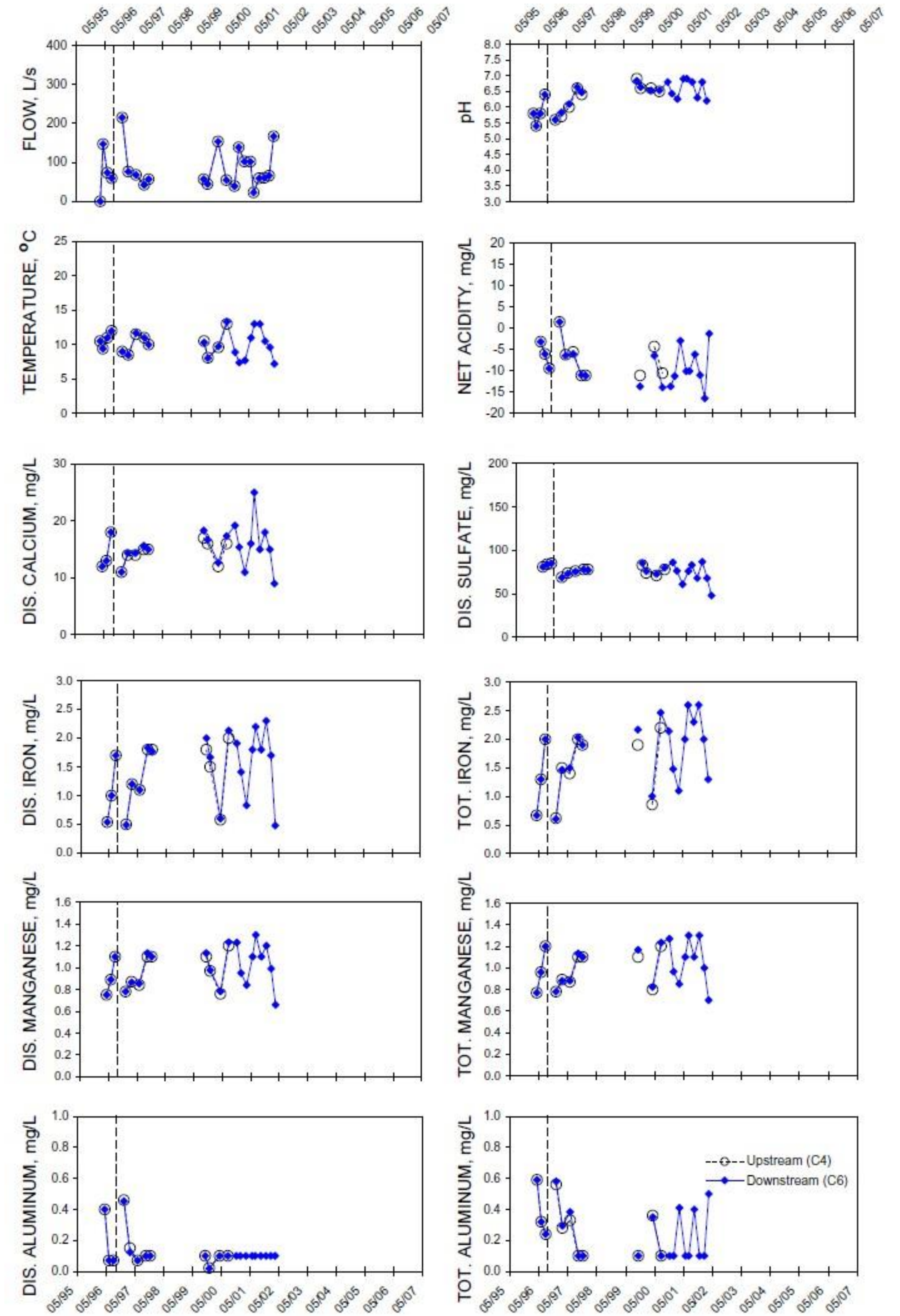

Figure 6. - Water-quality data upstream (C4) and downstream (C6) of treatment with limestone sand in Coal Run (LSC). Vertical dashed line indicates implementation date. Upstream data were not collected after July 2000.

Open Limestone Channel on Swatara Creek (OLS; B1 - B3). Before construction of the open limestone channel near the headwaters of Swatara Creek (Picture 1 and 2, Appendix A), the streamwater at sites B1 and B3 was acidic, with low pH (< 4.5), low concentrations of SO_4 (12 to

48 mg/L) and Mn (< 0.5 mg/L), and moderate concentrations of dissolved Al and Fe (0.5 to 3 mg/L) (Figure 7). These conditions and preliminary field experiments indicating initially rapid increases in the pH of streamwater in contact with limestone in an "open bucket" warranted the construction of the open limestone channel. The preliminary experiments indicated the rate of limestone dissolution decreased with increased pH, which is consistent with "open" cubitainer testing of nearby AMD (e.g. Cravotta 2003; Cravotta et al. 2004).

The first set of post-implementation samples indicated downstream increases in pH by about 1 unit equivalent to preliminary bucket tests (Figure 7). However, after only 2 months, an anoxic limestone drain was constructed on the Buck Mountain discharge at site A2. This anoxic limestone drain on an upstream tributary produced near-neutral streamwater at site B1 above the open limestone channel (Figure 7). The neutral upstream water was not aggressive toward limestone in the stream channel compared to initially acidic streamwater. Although concentrations of net acidity and the concentrations and loads of dissolved SO_4, Al, and Mn exhibited significant decreases, overall differences in pH and concentrations of dissolved Ca and Fe between the upstream site (B1) and downstream site (B3) at the open limestone channel were not significant for any flow conditions (Figure 5 and 7). The limestone channel treatment on Swatara Creek removed a median acidity of 0.2 t/y and added 0.1 t/y Ca as $CaCO_3$ over a 4-year monitoring period with paired samples (Figure 7, Table 3). Compared to the other seven treatments, the limestone channel on Swatara Creek had the lowest acid-removal efficiency (2.0 g/d/t) and the highest cost efficiency ($2,397/t) (Table 3).

Oxic Limestone Drain on Hegins discharge (ODH; H0 - H1). Before construction of the oxic limestone drain near the headwaters of Swatara Creek, AMD flowed from the Hegins discharge at site H0 for about 100 m as an unnamed tributary to site H1 at Swatara Creek (Figure 2). The oxic limestone drain was constructed as a series of limestone-filled cells within the channel of this unnamed tributary (Picture 7 and 8, Appendix A). The untreated AMD was oxic (DO 7.6 to 12 mg/L) and acidic (net acidity 22 to 65 mg/L) with low pH (3.3 to 4.2) and elevated concentrations of dissolved Al (2.7 to 6.4 mg/L) and Mn (0.9 to 2.5 mg/L) but moderate concentrations of dissolved Fe (< 1 mg/L) (Figures 4 and 8). The treated effluent had greater pH (4.0 to 6.9; median increase of 1.1) and smaller concentrations of net acidity and dissolved metals than the influent (Figure 8). The oxic limestone drain on the Hegins discharge removed a median acidity of 7.7 t/y and added 3.0 t/y Ca as $CaCO_3$ over the 7-year monitoring period (Figure 8, Table 3). Compared to the other seven treatment systems, the oxic limestone drain treatment had relatively low acid-removal efficiency (8.5 g/d/t) and moderately high cost efficiency ($891/t) (Table 3).

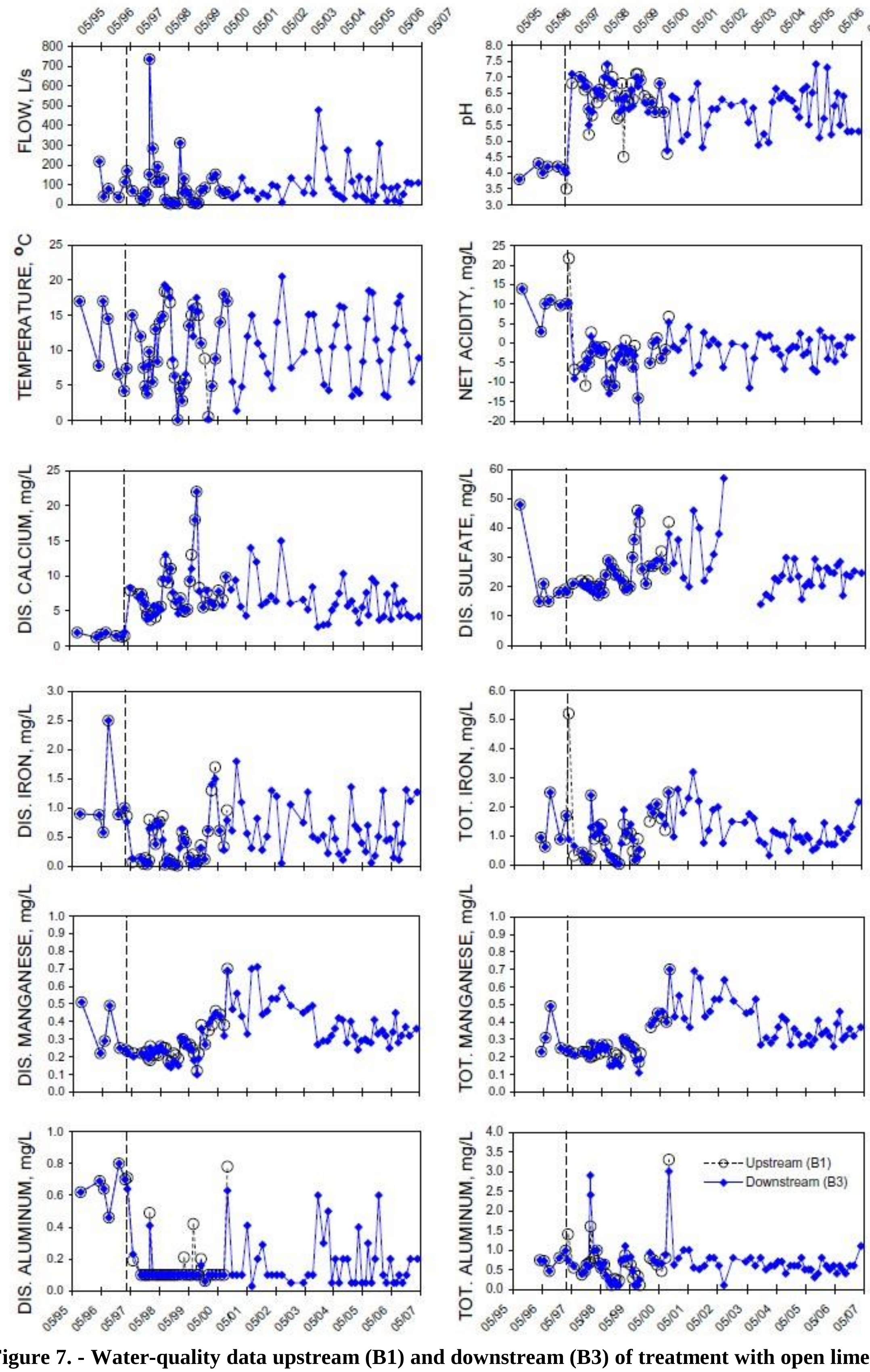

Figure 7. - Water-quality data upstream (B1) and downstream (B3) of treatment with open limestone channel on Swatara Creek (OLC). Vertical dashed line indicates implementation date. Upstream data were not collected after July 2000.

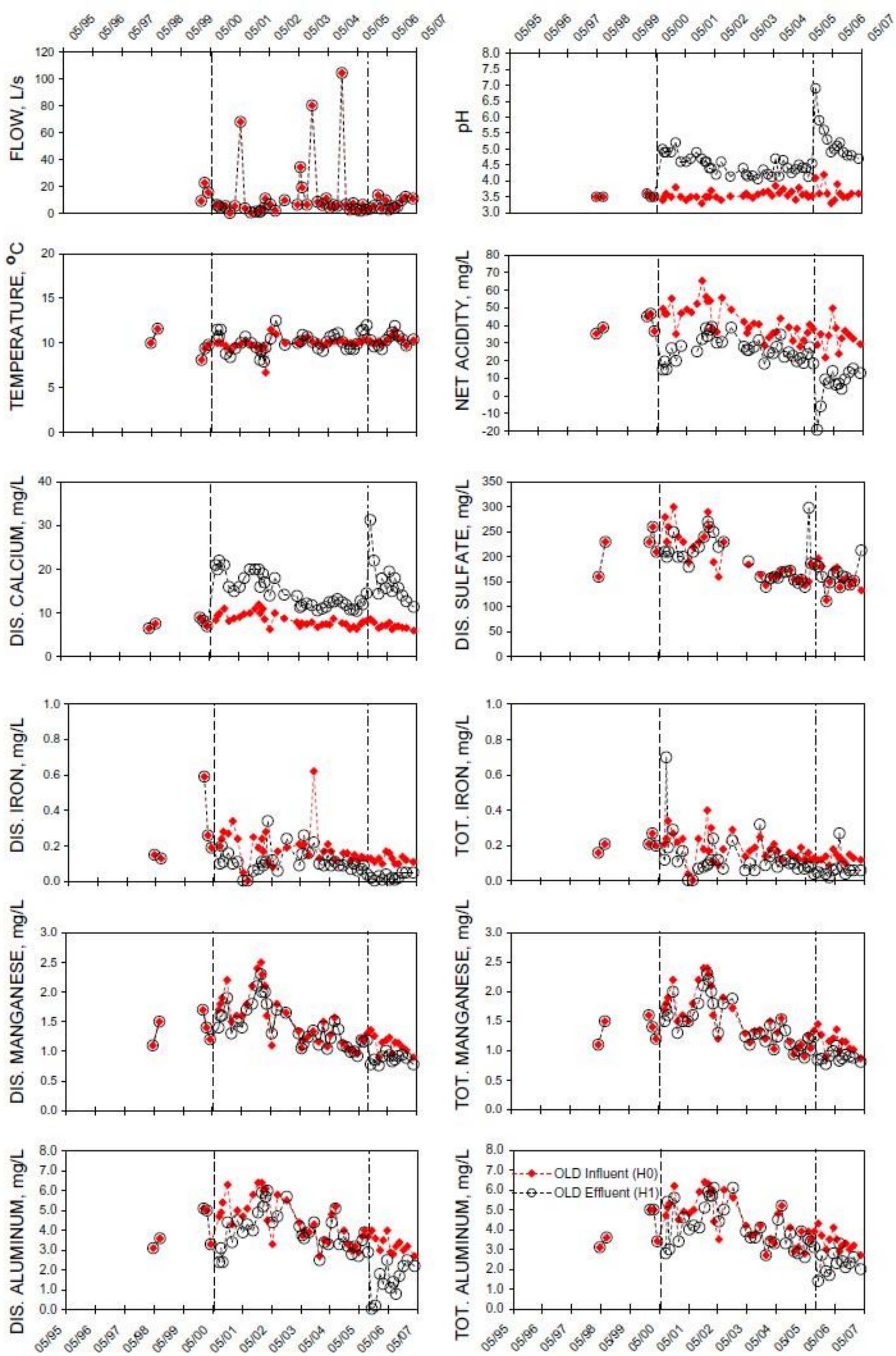

Figure 8. - Water-quality data upstream (H0) and downstream (H1) of treatment with oxic limestone drain at Hegins discharge (ODH). Vertical dashed line indicates implementation date. After initial implementation, limestone was added in September 2005 (dash-dot line).

The increased pH and concentrations of Ca and decreased concentrations of acidity, Fe, Mn, and Al were significant over the range of flow conditions (Figures 5 and 8). Nevertheless, the overall effectiveness of treatment improved with increased retention time, in conjunction with decreased flow rate as explained in more detail by Cravotta et al. (2004). After the system was enlarged with additional limestone and covered with compost in September 2005, the $CaCO_3$ load increased and the treatment effectiveness improved (Figure 8). The larger size of the oxic limestone drain resulted in an increased retention time, and the added compost helped increase the partial pressure of CO_2, both of which enhanced limestone dissolution within the treatment cells.

Anoxic Limestone Drain on Buck Mountain discharge (ADB; A1 - A3). Before construction of the anoxic limestone drain on the Buck Mountain discharge (Picture 5 and 6, Appendix A), AMD flowed from the Buck Mountain discharge at site A1 for about 550 m as an unnamed tributary to site A3 near the headwaters of Swatara Creek (Figure 2). The untreated AMD generally was suboxic (DO < 2.0 mg/L) with slightly acidic pH (4.1 to 6.1) and elevated concentrations of dissolved Fe (4 to 24 mg/L) but moderate concentrations of dissolved Al ($\leq$ 1.2 mg/L) and Mn ($\leq$ 2.3 mg/L) (Figure 4 and 9). As this untreated effluent flowed downstream to site A3, the pH and concentrations of dissolved Fe decreased due to oxidation and hydrolysis reactions; however, dissolved Al remained elevated (> 0.3 mg/L) (Figure 9).

After implementation of the anoxic limestone drain, the downstream water at site A3 was maintained with near-neutral pH and positive net alkalinity (net acidity < 0) (Figure 9). The anoxic limestone drain effluent at site A2 had significantly greater pH (median increase 1.6), greater concentrations and loads of Ca, and smaller concentrations and loads of net acidity and dissolved Fe and Al than the influent at A1 (Figures 5 and 9). These effluent characteristics were consistent with results of cubitainer tests with the AMD from the Buck Mountain discharge (Cravotta 2003; Cravotta et al. 2004). In contrast with pretreatment conditions, the pH of treated effluent increased downstream to site A3 due to the exsolution of CO_2 (Figure 9). Because of additional inflows of diffuse AMD along the tributary, the flow rate, net acidity, and SO_4 concentrations also increased downstream. Nevertheless, alkalinity added by the anoxic limestone drain was sufficient to buffer the downstream pH at site A3.

The anoxic limestone drain decreased the acidity by a median of 26.2 t/y and increased the dissolved Ca by a median of 27.9 t/y as $CaCO_3$ (Table 3). This added $CaCO_3$ load was substantially larger than that from other limestone treatment systems in the watershed and indicates a rapid rate of limestone dissolution in the anoxic limestone drain as explained in detail by Cravotta (2003) and Cravotta et al. (2004). Because of the rapid rate of limestone consumption, the Buck Mountain anoxic limestone drain, which had an original size of 320 t, was enlarged during the study with the addition of 100 t of limestone in January 2001 and again in September 2005. Including these enlargements, the anoxic limestone drain treatment had relatively high acid-removal efficiency (52.1 g/d/t) and a low cost efficiency ($131/t) compared to the other treatment systems (Table 3).

The anoxic limestone drain was effective for neutralization of AMD and attenuation of dissolved metals over the range of flow conditions but was not always effective for attenuation of total metals. Specifically, during low-flow conditions, the concentration of total Al decreased downstream from site A1 to site A3 (Figure 9). However, sporadically during normal to high-flow conditions, the concentration and load of total Al increased downstream (Figure 9). During normal to high flows, turbulent water could transport freshly precipitated Al-hydroxide particles downstream.

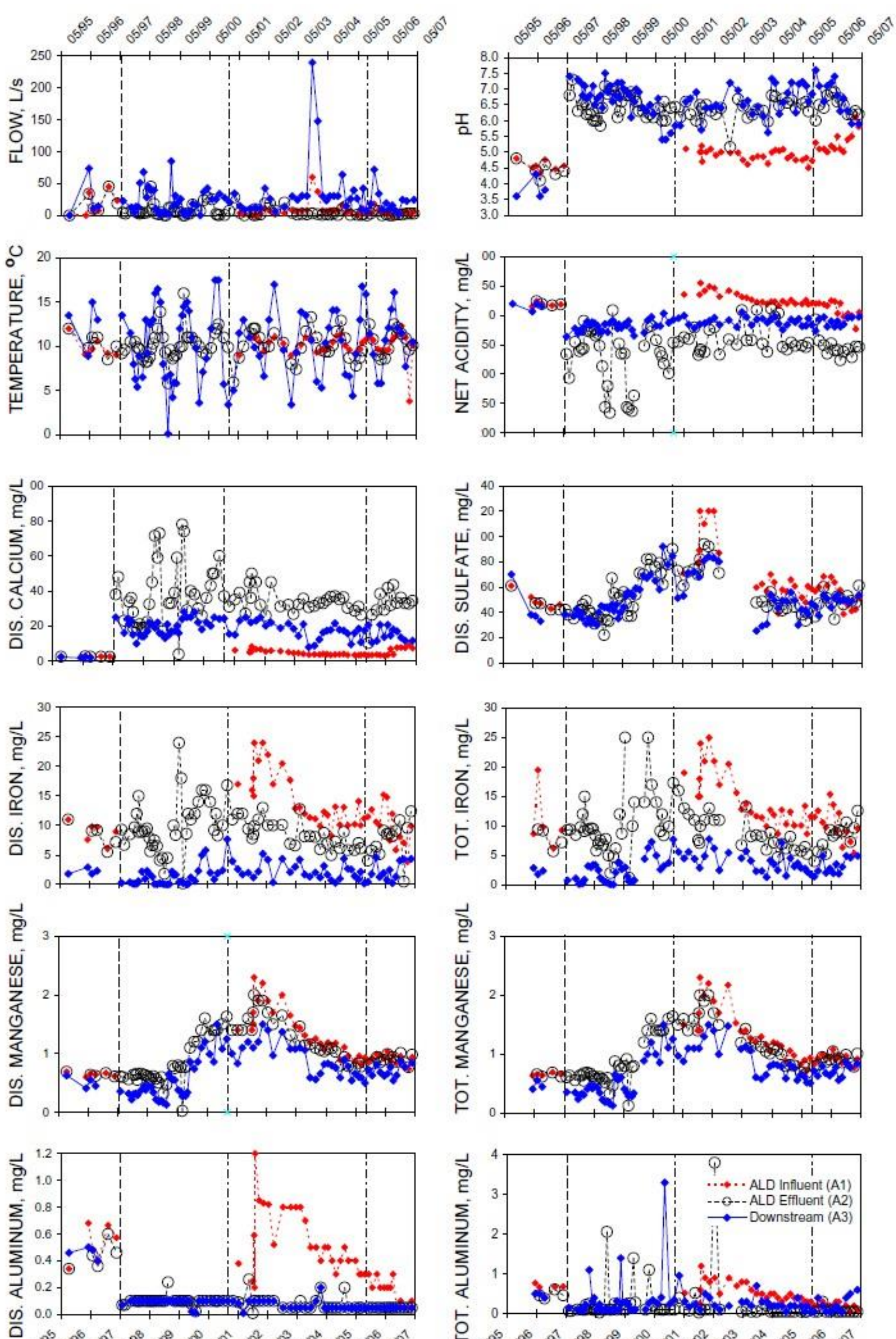

Figure 9. Water-quality data upstream (A1) and downstream (A2, A3) of treatment with anoxic limestone drain at Buck Mountain discharge (ADB). Vertical dashed line indicates implementation date. After initial implementation, limestone was added in January 2001 and September 2005 (dash-dot line).

Limestone Diversion Wells on Swatara Creek (DWS; C1 - C3). Before installation of the two limestone diversion wells near the headwaters of Swatara Creek (Picture 9, Appendix A), the streamwater at sites C1 and C3 had the same values of pH < 4.5, dissolved Al > 1.5 mg/L, and associated constituents (Figure 10). After implementation of the treatment, the pH of streamwater at the downstream site (C3), approximately 140 m below the diversion wells, typically was 0.5 units higher than that at the upstream site (C1) (Figure 5 and 10). The median decrease in net acidity load was 9.1 t/y, and the increase in Ca load was 8.6 t/y as $CaCO_3$. Assuming limestone consumption at a rate of 30 t/y for each diversion well, the treatment on Swatara Creek had relatively low acid-removal efficiency (12.7 g/d/t) and high cost efficiency ($1,035/t) compared to the other treatment systems (Table 3).

For most flow conditions, the limestone diversion wells increased pH and concentrations of Ca and decreased concentrations of net acidity and dissolved Fe and Al (Figures 5 and 10). Nevertheless, during extreme high-flow conditions associated with tropical storms in September 1999 and spring storms in March-May 2000, the pH of downstream water (continuously monitored) was not effectively increased (Cravotta and Weitzel 2001). During such stormflow conditions, the effectiveness of the limestone diversion wells was diminished because a smaller proportion of total streamflow was treated.

Limestone Diversion Wells on Lorberry Creek (DWL; E2-0 - E2-1). Below the Rowe Drainage Tunnel, Lorberry Creek at site E2-0 had extremely variable pH (3.9 to 6.5) and net acidity (-2.2 to 54 mg/L as $CaCO_3$) and elevated concentrations of dissolved Fe (2.0 to 12 mg/L) and Al (0.01 to 5.8 mg/L) (Figures 4 and 11). Although the diversion wells on Lorberry Creek below Rowe Drainage Tunnel (Picture 10, Appendix A) did not treat the entire flow of the Rowe Tunnel, they effectively increased pH and decreased net acidity and dissolved Fe and Al concentrations in the downstream segment of Lorberry Creek at site E2-1 over a wide range of flow conditions (Figures 5 and 11). The median decrease in net acidity load was 40.9 t/y, but the increase in Ca load was only 1.3 t/y as $CaCO_3$. Assuming limestone consumption at a rate of 30 t/y for each diversion well, the treatment on Lorberry Creek had very high acid-removal efficiency (75.8 g/d/t) and relatively low cost efficiency ($230/t) compared to the other treatment systems (Table 3). Nevertheless, the large inconsistency between the acid load removed and the $CaCO_3$ load added (Table 3) could indicate a source of alkalinity other than the limestone diversion wells between the upstream and downstream monitoring sites, such as sodium carbonate from an abandoned treatment tank below the upstream monitoring site. Using the $CaCO_3$ load added as a surrogate for the acid removal associated with the limestone diversion wells, the removal efficiency decreases to 2.4 g/d/t and the cost efficiency increases to $7,377/t, which would be among the least efficient of the treatment systems.

The limestone diversion wells on Lorberry Creek added alkalinity and increased pH over most flow conditions. Nevertheless, during extreme high-flow conditions associated with large storms, the pH was changed little (Cravotta and Weitzel 2001) and the dissolved Ca concentration decreased downstream of the diversion wells on Lorberry Creek. During high-flow conditions, a large fraction of the AMD bypassed the diversion wells and the treated effluent could be diluted by runoff or groundwater seepage between the upstream and downstream monitoring sites.

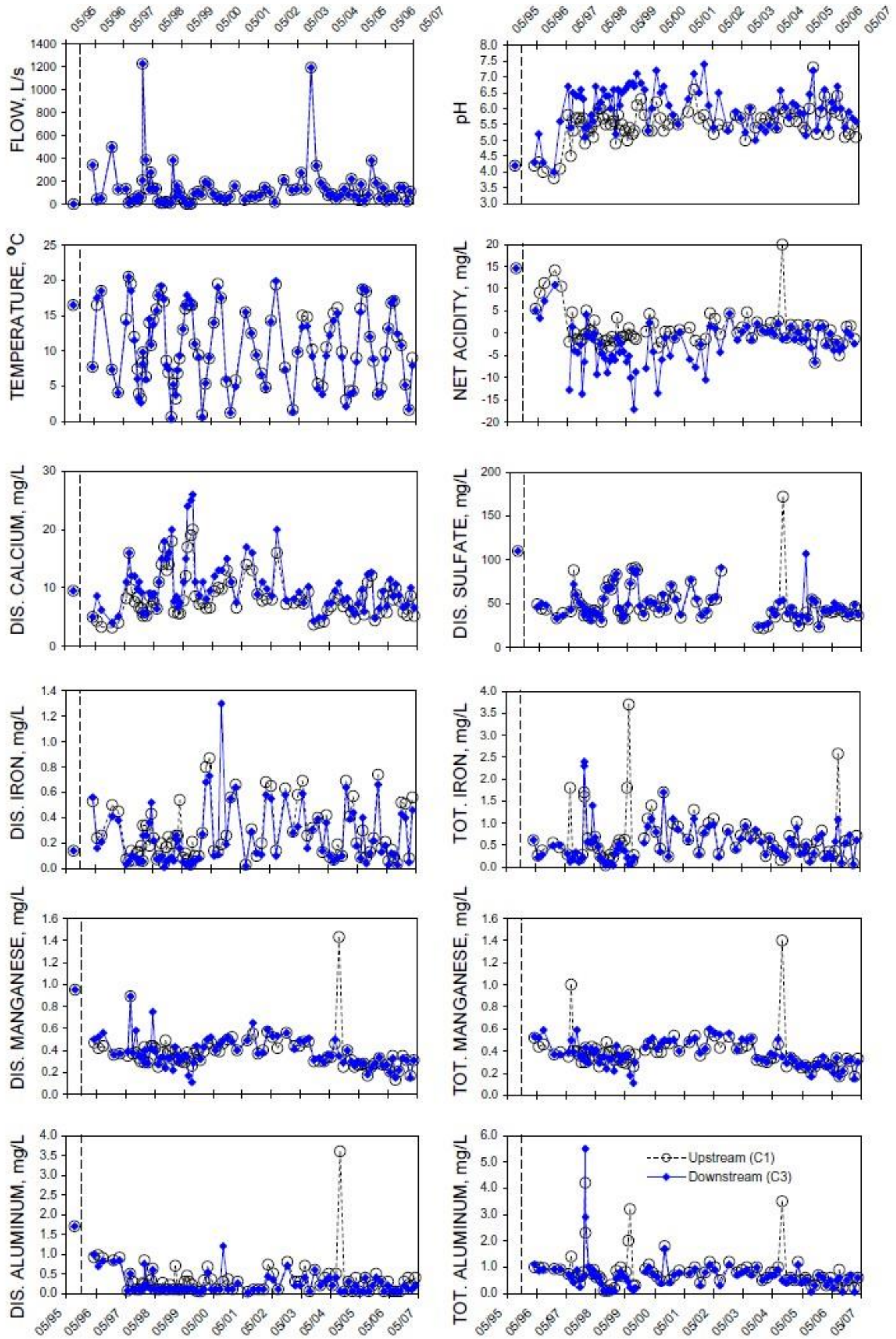

Figure 10. Water-quality data upstream (C1) and downstream (C3) of treatment with limestone diversion wells on Swatara Creek (DWS) near Newtown. Vertical dashed line indicates implementation date.

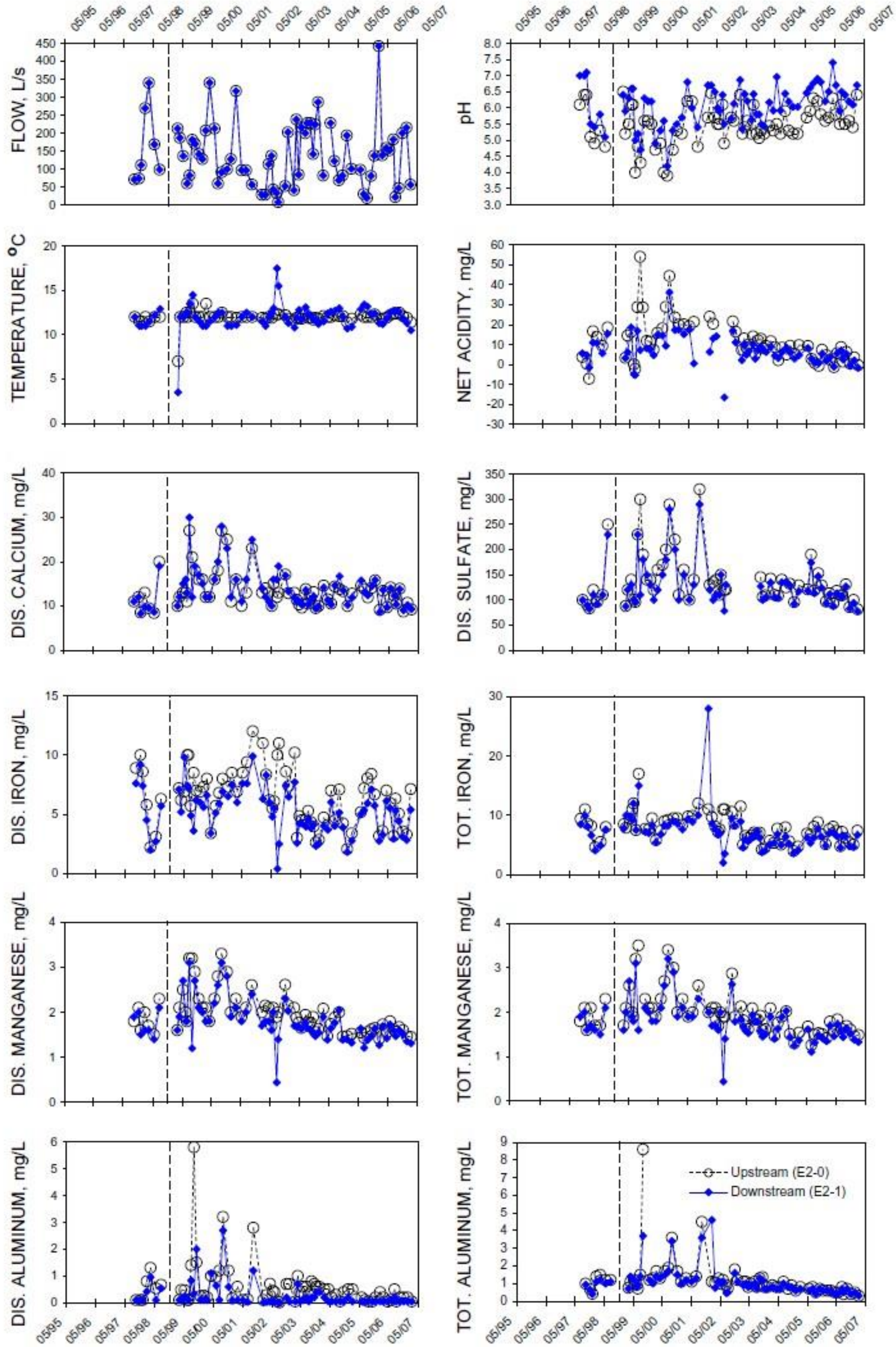

Figure 11. - Water-quality data upstream (E2-0) and downstream (E2-1) of treatment with limestone diversion wells on Lorberry Creek (WLL) below the Rowe Tunnel discharge. Vertical dashed line indicates implementation date.

Aerobic Wetlands below Diversion Wells on Lorberry Creek (WLL; E2-1A - E2-3). Before installation of the aerobic wetlands downstream from the diversion wells on Lorberry Creek (Picture 10, Appendix A), the effluent from the diversion wells was discharged directly to the stream where the increased pH from treatment promoted the precipitation of Fe and Al solids in the stream channel. The wetlands were constructed to remove the metals from the effluent by providing a location for the oxidation of Fe and settling of metal-rich solids (Picture 12, Appendix A).

As anticipated, the wetlands on Lorberry Creek increased pH and decreased concentrations of net acidity and dissolved and total Fe and Al over a wide range of flow conditions (Figures 5 and 12). Concentrations of dissolved Ca and Mn increased within the wetlands because of continuous groundwater seepage into the wetlands and the irregular addition of hydrated lime to the influent (a lime doser was rarely in service). The median decrease in net acidity load was 12.3 t/y, and the increase in Ca load was 23.8 t/y as $CaCO_3$. Ignoring the quantities and cost of the added lime, the wetland treatment on Lorberry Creek had very low acid-removal efficiency (2.5 g/d/t) and relatively high cost efficiency ($1,584/t) compared to the other treatment systems (Table 3). If the cost for lime was considered, the treatment cost efficiency would be even greater.

Although the Lorberry wetlands effectively removed Fe and Al from Lorberry Creek, they had another unintended effect. Instead of sustaining a year-round water temperature of approximately 13^{o}C exhibited by the Rowe Tunnel discharge and the Lorberry Creek diversion well effluent (Figure 11), the temperature of the wetland effluent ranged widely (Figures 5 and 12). During summer, the temperature of the wetland effluent increased to 25.4^{o}C, which greatly exceeds the upper limit of 18.7^{o}C for a cold-water fishery (Commonwealth of Pennsylvania 2002). Although brook trout were reported in the lower reaches of Lorberry Creek during the study period (Cravotta 2009), the potential for adverse temperature effects should be considered with possible plans for additional wetlands to treat other AMD sources, such as the Shadle or Pantherhead discharges (Figure 2).

Limestone-Compost-Based Wetlands on Lower Rausch Creek (WLR; E3-1 - E3-2). The wetlands constructed on Lower Rausch Creek impounded streamflow within a highway fill area that was underlain by boulders and was prone to losing water (Picture 11, Appendix A). Hence, because of seepage losses and evaporative losses, the flow rate exiting the wetlands typically was less than that entering the wetlands (Figure 5). If the water losses were solely from evaporation, the dissolved chemical concentrations could increase within the wetlands, whereas the loads of relatively conservative solutes, such as SO_4, would not be affected. The upstream water entering the Rausch Creek wetlands had pH of 6.2 to 7.8 and concentrations of net acidity of -24.8 to 2.3 mg/L, dissolved SO_4 of 62 to 207 mg/L, and widely variable concentrations of dissolved Fe, Al, and Mn ranging from < 0.05 to > 1.5 mg/L (Figure 13). In comparison, the water downstream from the wetlands exhibited significantly lower pH and smaller concentrations and loads of net acidity, dissolved SO_4, and dissolved and total Fe, Al, and Mn (Figure 5). Although the concentrations of dissolved and total Ca were equivalent for the influent and effluent, the load of Ca decreased through the wetlands (Figure 13).

The Rausch Creek wetlands removed a median acidity load of 3.1 t/y (Table 4). In contrast with the other treatment systems that exported Ca, the Rausch Creek wetlands removed 27.0 t/y Ca as $CaCO_3$, which resulted from flow losses through the wetlands. Compared to the other treatment systems, the wetland treatment on Rausch Creek had the lowest acid-removal efficiency (0.3 g/d/t) and the highest cost efficiency ($7,783/t) (Table 4). Considering that flow losses magnified the apparent acid-removal rates, the actual treatment efficiency would be worse than indicated.
Despite flow losses, chemical reactions could have caused a decrease in the concentrations and loads of dissolved SO_4 and dissolved Fe, Al, and Mn within the Rausch Creek wetlands (Figure 13). Negative values of the saturation index for gypsum ($CaSO_4 \cdot 2H_2O$; SI $\leq$ -1.35) indicate that precipitation of SO_4 from the effluent would not have been a feasible mechanism for its removal. In contrast, the effluent typically was near saturation or supersaturated with respect to $Fe(OH)_3$ and $Al(OH)_3$, indicating potential for removal of dissolved Fe and Al by precipitation of such

phases. Although the water column was presumed aerobic, reducing conditions could have developed in the underlying compost. The implication is that multiple processes such as Fe oxidation and settling within the water column and dissimilatory SO_4 reduction within the compost substrate may have been active in the wetlands. Because the median pH decreased within the wetlands, alkalinity produced by SO_4 reduction (if active) was not sufficient to buffer acidity released by any such Fe^{III} or Al hydrolysis reactions. Furthermore, although most pairs of samples indicated declines in metal concentrations and transport from the upstream to downstream monitoring sites, three pairs collected during different stormflow conditions indicated concentrations of total metals and suspended solids were greater at the downstream site than the upstream site for the Rausch Creek wetlands. Consequently, the wetlands could export metals during high-flow conditions.

As described for the Lorberry Creek wetlands, the temperature of the Rausch Creek wetland effluent increased to 24°C during summer months, which exceeds the upper limit of 18.7°C for a cold-water fishery (Commonwealth of Pennsylvania 2002). Although seepage losses from the Rausch Creek wetland were unintended and were not monitored directly, a treatment system designed to transmit wetland effluent through the subsurface before discharging to the stream could reduce the effect of temperature variations resulting from impoundment of water within wetlands (e.g. Cravotta 2007).

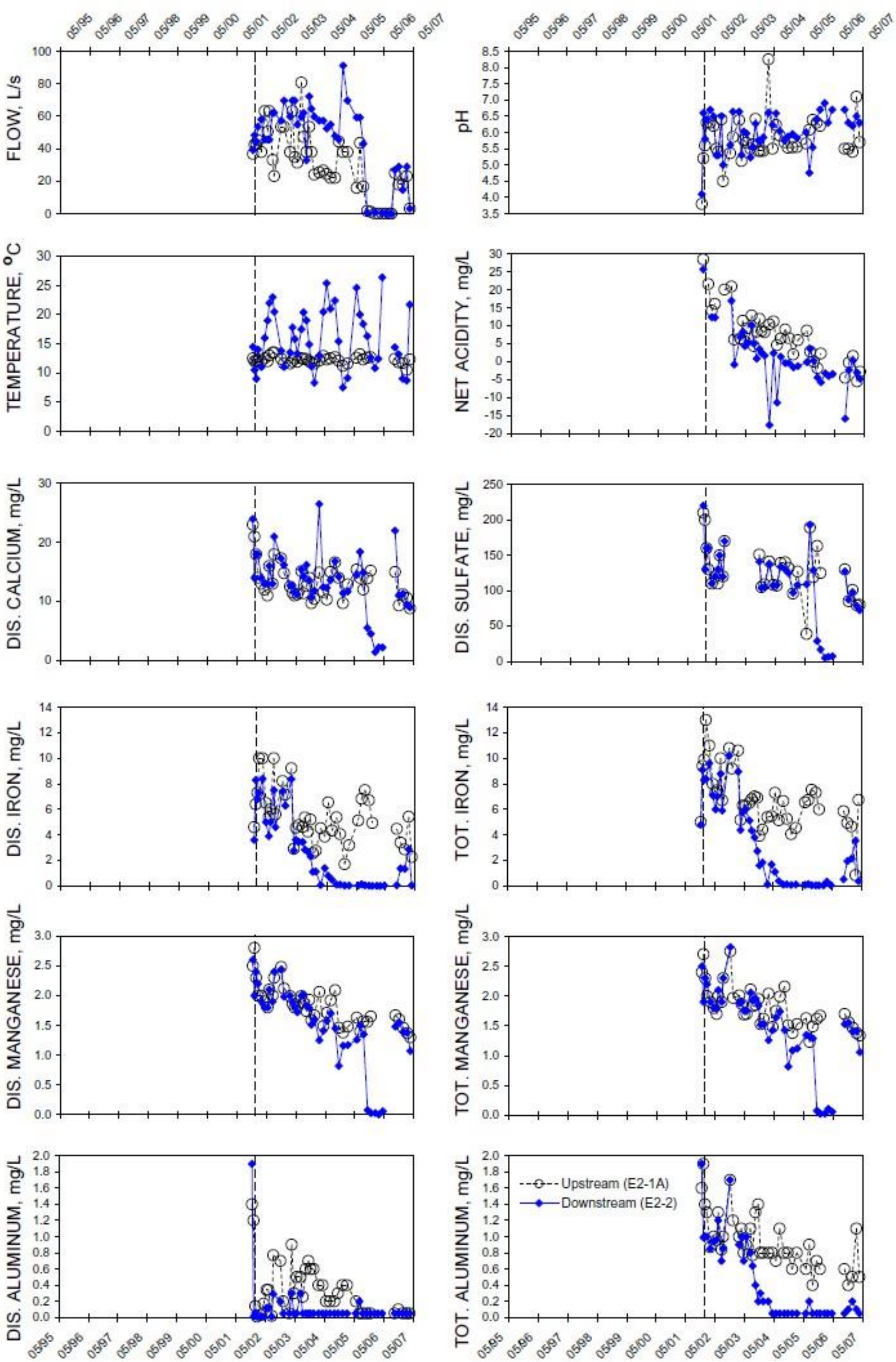

Figure 12. - Water-quality data upstream (E2-1A) and downstream (E2-2) of treatment with aerobic wetlands on Lorberry Creek (WLL) below the diversion wells. Vertical dashed line indicates implementation date.

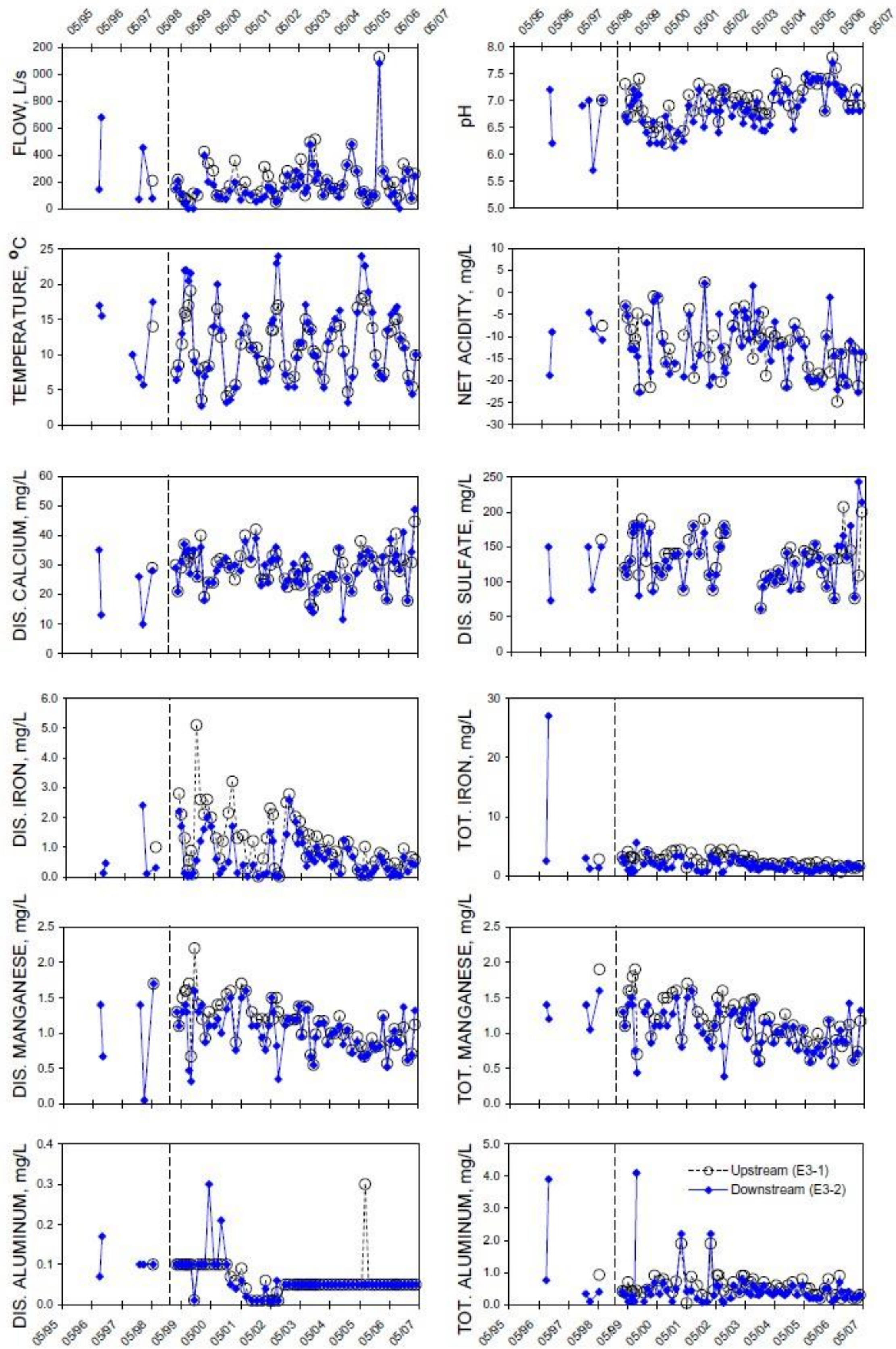

Figure 13. - Water-quality data upstream (E3-1) and downstream (E3-2) of treatment with limestone- compost wetlands on Lower Rausch Creek (WLR). Vertical dashed line indicates implementation date.

Comparing Streamwater Quality with Streamflow

Temporal variability in streamflow is one of the most important factors affecting water quality. Although annual streamflow was within the normal range during 1996-1998 and 2005-2007, it was lower than normal in 1999-2001 and greater than normal in 2002-2004 (Figures 14A and 15A). Hydrograph separation with PART (Rutledge 1998) indicated the total streamflow at Ravine during the study was composed of about 75 % base flow and 25 % storm runoff (Table 5). Generally the runoff associated with stormflow events lasted from hours to several days.

Expressed as the yield, where streamflow is divided by drainage area, the upstream station on Swatara Creek at Newtown had lower annual mean streamflow than downstream gaging stations on Swatara Creek at Ravine, Pine Grove, and Harper Tavern (Table 5). Relatively small streamflow yields for Swatara Creek at Newtown are consistent with this drainage area losing water to the underground mine that flows eastward to the Otto Colliery in the adjacent watershed (Gannett Fleming Corddry and Carpenter, Inc. 1972). In contrast, large streamflow and base-flow yields for Lorberry Creek (Table 5) are consistent with groundwater inflows from outside the delineated watershed. During the present study and historically, the Rowe Tunnel discharge accounted for more than 60 % of the annual streamflow of Lorberry Creek (Anthracite Research and Development Co., Inc. 1972). The Rowe Tunnel drains the Lincoln Mine pool that extends beneath the Lorberry Creek and Lower Rausch Creek watersheds (Anthracite Research and Development Co., Inc. 1972).

At Ravine, the continuously recorded pH ranged from 4.7 to 8.2 and SC ranged from 27 to 540 μS/cm during the study (Figures 14 and 16); pH and SC values generally decreased with increased streamflow (Figure 16). Minimum values of pH and SC were recorded for stormflow, implying that storm runoff that mixes with base flow is both acidic and dilute, as explained in more detail below.

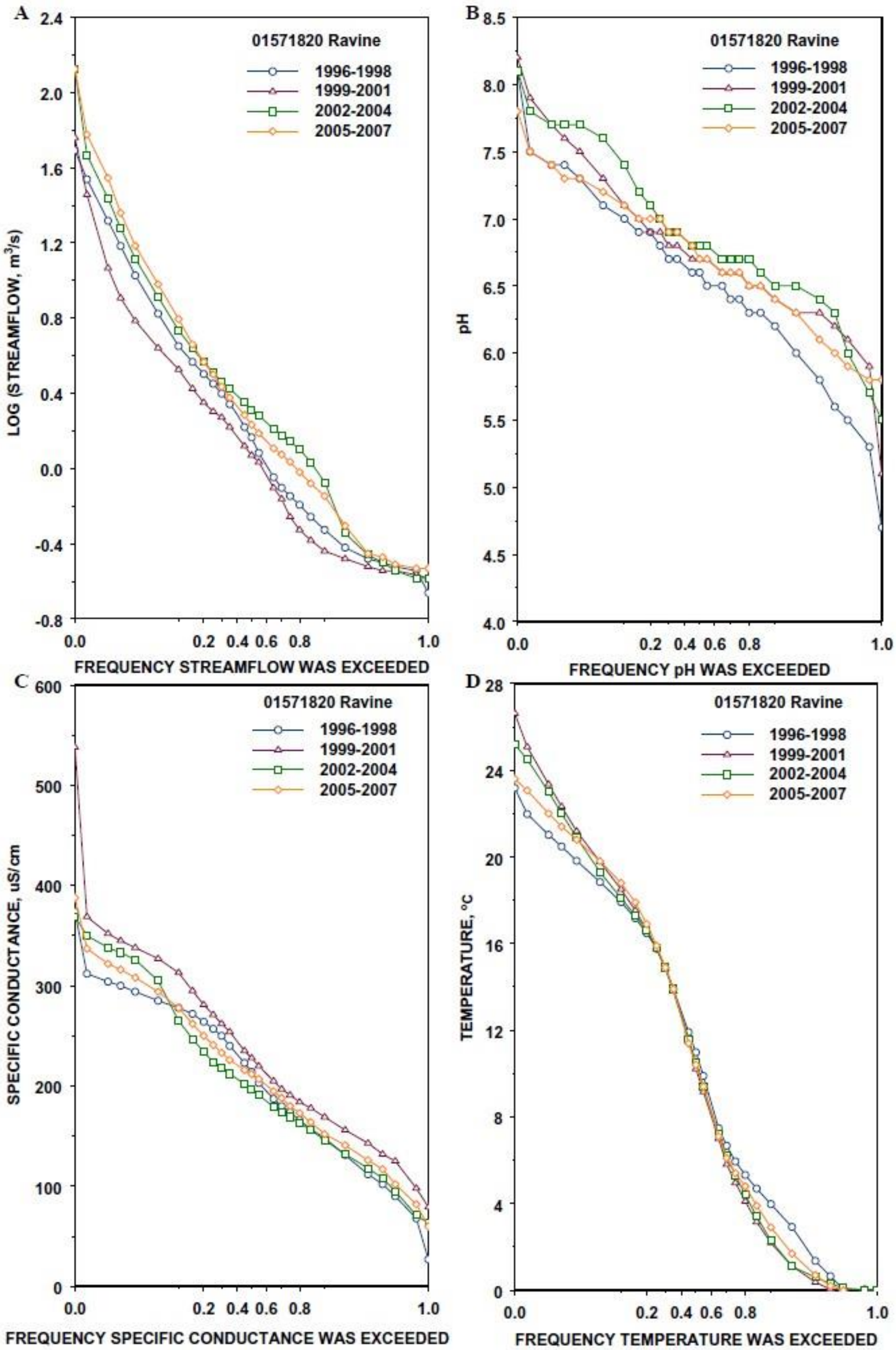

Figure 14. Probability plots of continuously measured (recorded at 15-minute intervals) data for Swatara Creek at Ravine, Pa., September 1996 through September 2007: A, streamflow; B, pH; C, specific conductance; D, temperature. The X-axis indicates the frequency that values were exceeded during 3- year intervals.

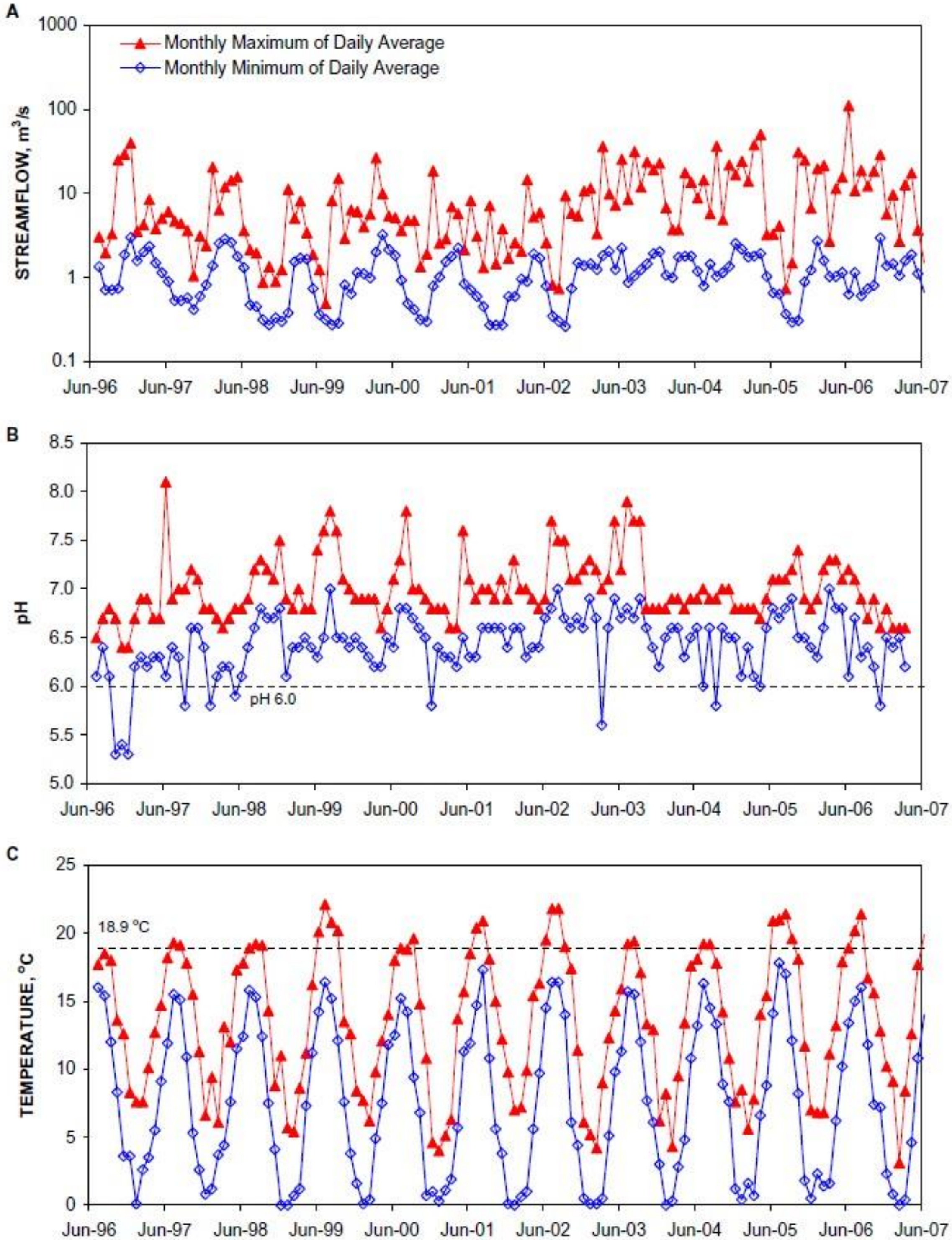

Figure 15. Time-series plots of the monthly range (maximum and minimum) of daily mean values for Swatara Creek at Ravine, Pa., June 1996 through June 2007: A, streamflow; B, pH; and C, temperature. In B and C, dashed horizontal lines indicate minimum pH and maximum temperature permitted for "cold-water fishery" in July and August, respectively (Commonwealth of Pennsylvania, 2002).

Table 5. Hydrograph-separation analysis[12] and components of the annual hydrologic budget for streamflow-gaging stations in the upper Swatara Creek Basin
[km^2, square kilometers; m^3/s, cubic meters per second; cm/yr, centimeters per year; %, percent]

US Geological Survey Station ID	Gage Location	Drain-age Area, km^2	Time Period[13]	Mean Streamflow[14]		Mean Base Flow[15]			Mean Runoff[16]	
				m^3/s	cm/yr	m^3/s	cm/yr	Index, %	m^3/s	cm/yr
01573000	Swatara Creek at Harper Tavern	862.7	1920-2006	16.44	60.1	10.12	37.0	61.5	6.32	23.1
01573000	Swatara Creek at Harper Tavern	862.7	1997-2006	17.33	63.4	10.55	38.6	60.9	6.78	24.8
01572025	Swatara Creek at Pine Grove	297.0	1997-2006	6.13	65.1	4.27	45.4	69.7	1.86	19.8
01571820	Swatara Creek at Ravine	110.8	1997-2006	2.44	69.5	1.84	52.4	75.4	0.60	17.1
0157155014	Swatara Creek at Newtown	7.5	1997-2006	0.13	54.0	0.10	42.1	78.0	0.03	12.6
01573000	Swatara Creek at Harper Tavern	862.7	2000-2006	18.83	68.9	11.08	40.5	58.8	7.75	28.3
01572025	Swatara Creek at Pine Grove	297.0	2000-2006	6.58	69.9	4.48	47.6	68.1	2.1	22.3
01571820	Swatara Creek at Ravine	110.8	2000-2006	2.64	75.2	1.95	55.5	73.8	0.69	19.7
0157155014	Swatara Creek at Newtown	7.5	2000-2006	0.14	57.4	0.11	46.3	80.6	0.03	12.6
01571778	Lorberry Creek at Lorberry Jct	9.0	2000-2006	0.31	109.1	0.27	94.7	86.8	0.04	14.0

a Hydrograph separation was conducted using the "PART" computer program (Rutledge 1998) to divide annual streamflow into base flow (B) and runoff (R) contributions on the basis of daily mean streamflow values during time period indicated.

b Time period is the range of water years, from October of the prior calendar year through September of the calendar year.

c Streamflow expressed as centimeters per year by dividing streamflow in cubic meters per second by the drainage area in square kilometers and then multiplying by the factor 3,156.

e Base flow expressed as cubic meters per second, centimeters per year, and percent of total annual streamflow (base-flow index).

f Runoff expressed as cubic meters per second or centimeters per year was computed by subtracting the base flow from total streamflow.

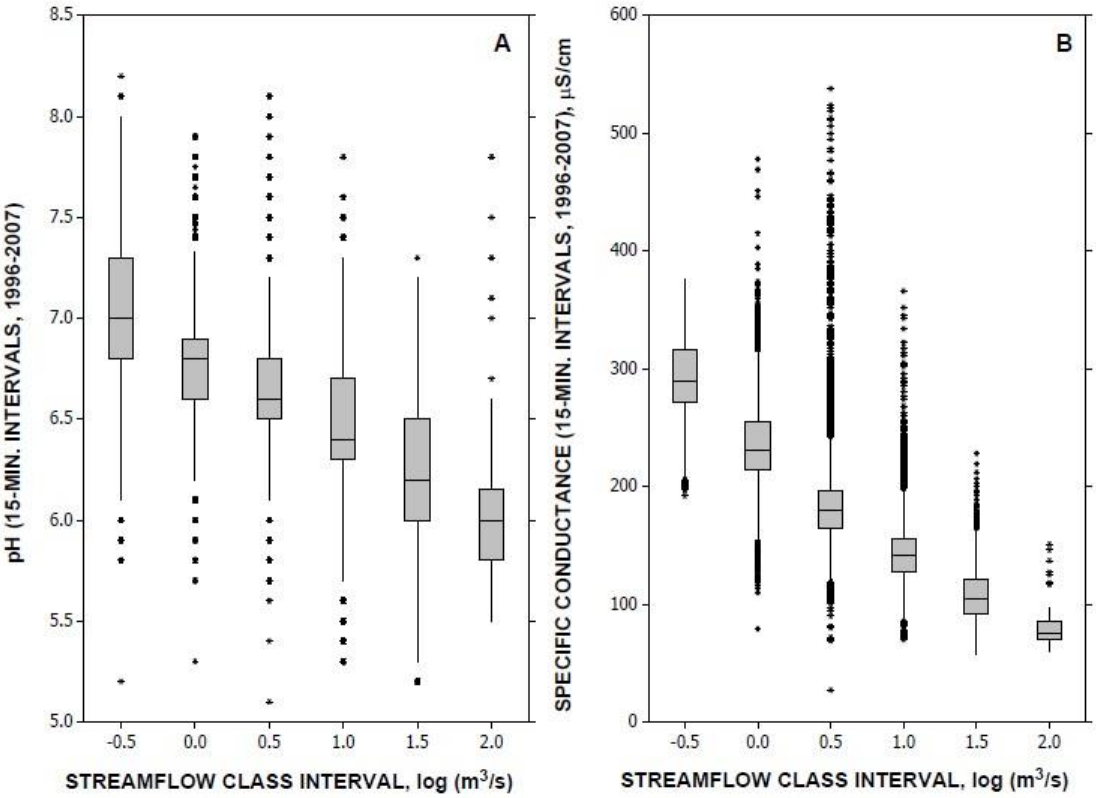

Figure 16. Boxplots showing continuously measured (recorded at 15-minute intervals) pH and specific conductance as a function of streamflow for Swatara Creek at Ravine, Pa., September 1996 through September 2007. Each streamflow class interval on the x-axis, where numbers are logged values of streamflow in cubic meters per second, includes values within 0.25 of the listed value (e.g. 1.0 is 0.75 to 1.25). Shaded area of box indicates the "interquartile" range (IQR = 25th to 75th percentile); horizontal line inside the box indicates the median; vertical lines extend to extreme values within 1.5 times the IQR; symbols indicate outlier values.

One could hypothesize that with the implementation of limestone-based treatment systems at many of the AMD sources in the Swatara Creek Basin during the late 1990s (Figure 2), streamflow would not be affected, but pH, alkalinity, and calcium concentrations would increase at downstream sites. Frequency distribution plots of continuous-record streamflow, pH, SC, and temperature of Swatara Creek at Ravine for 3-year intervals during 1996-2007 (Figure 14) show that the streamflow distribution during 1996-1998 was comparable to the long-term distribution. However, during the 1996-1998 period, Swatara Creek had a greater frequency of low values of pH and SC and a smaller range in temperature compared to later periods (Figures 14 and 15). The decrease in the frequency of low values of pH and SC and the increase in the range of temperature after 1998 coincides with, and could result from, the implementation of AMD treatments. Limestone diversion wells, limestone drains, and limestone channels are sources of dissolved solids (as calcium and alkalinity) that would tend to increase the pH and SC. Constructed wetlands and the diversion of streams from mines to surface channels would have little effect on dissolved solute concentrations but could affect water temperature. After 1998, maximum stream temperatures increased during summer months and decreased during winter months (Figure 15). This increased range in maximum temperature is consistent with increased thermal exchange with the ambient atmosphere that could result from the impoundment of AMD in wetlands and the restoration of streamflow at mine-infiltration sites. Evaporation of stream water during low-flow periods would tend to amplify these effects on temperature and SC.

During 1996-2007, stream-water-quality samples for chemical analysis were collected for a wide range of hydrologic conditions (Figure 17). The samples collected with automated sampling devices during storm events were identified as rising, peak, and falling "stormflow" samples on the basis of the hydrograph for Swatara Creek at Ravine on the date of sampling. Samples collected during relatively stable stream stage between storm events were characterized as normal, low, and high “base-flow” samples.

Base flow. Current and historical data from 1959 to 2007 for Swatara Creek at Ravine indicate progressive improvement in streamwater quality (Figure 18). Although streamflow at times of collection of historical (1959-1985) and current (1996-2007) base-flow samples generally was comparable, sulfate decreased from a median of about 150 mg/L in 1959 to 75 mg/L in 2007; pH increased sharply from 3.5-4.4 (median ~4) to 4.6-7.0 (median ~6.5) after 1995 (Figure 18). Concentrations of dissolved iron and aluminum generally decreased with increased pH. The decrease in concentrations of sulfate and associated AMD contaminants in Swatara Creek over the past 50 years could result from a progressive decrease in contaminant loading from AMD sources after the initial flooding of the abandoned mines. Flooding of a mine can result in the (1) dissolution of accumulated pyrite-oxidation products, (2) reduction in the access of oxygen to the subsurface with a corresponding decrease in the pyrite oxidation rate, and (3) progressive dilution of initially acidic water by alkaline groundwater inflows. Such processes could account for gradual improvement in AMD and surface-water quality that has been ongoing for decades throughout the region, particularly in the Northern, Western, and Southern Anthracite Coalfields (e.g. Wood 1996). The associated increase in pH of Swatara Creek probably was caused by the onset of carbonate buffering that occurred when the rate of alkalinity production equaled or exceeded acid production (e.g. Cravotta et al. 1999). The implementation of limestone-based treatment systems during 1995-2001 would be expected to enhance the potential for carbonate buffering.

Stormflow. Storm-runoff events can occur year round in the study area and can have a dramatic effect on streamflow. Generally, monthly runoff as a fraction of total streamflow was greatest during the late summer and early fall, when seasonal low base flow typically was punctuated by large storms of tropical origin. Expressed as a percentage of monthly-total streamflow at Ravine, the annual mean base-flow and runoff fractions were 75 % and 25 % (Table 5). However, during typical low-base-flow conditions in late summer and early fall, a large percentage of the streamflow was “storm runoff” estimated with PART (Rutledge 1998). Months with an exceptionally high fraction of stormflow during the study included October 1996 (47 %), September 1999 (58 %), September 2001 (46 %), October 2003 (45 %), September 2004 (70 %), and October 2005 (52 %). In conjunction with large storm events, runoff was estimated to contribute 70 to 99 % of the daily mean streamflow during October 19-21, 1996; September 16-17, 1999; September 30-October 1, 1999; September 24-26, 2001; October 27-29, 2003; September 18-21, 2004; October 7-9, 2005; and September 2-4, 2006 (Figure 19).

Several examples of storm hydrographs during September and October 1996-2006 with associated stream chemistry are illustrated in Figure 17. With one exception, the same vertical axes for streamflow, SC, and sulfate; pH; and concentrations of suspended solids, total iron, and dissolved iron were used so that storm characteristics can be compared. Although each storm hydrograph is unique, owing to variations in storm duration, intensity, and runoff contribution, some features are consistent among the hydrographs. Specifically, as streamflow increased during storm events, the pH, SC, and sulfate concentration decreased, whereas the concentrations of suspended solids and total and dissolved iron increased (Figure 19). Other sampled hydrographs for all months of the year exhibited comparable patterns, except that storm events during 1996-1998 exhibited greater propensity for change in pH, with lower extremes (Figure 14B), than later years.

Six bulk precipitation samples were collected during June 1999 - June 2000 at Ravine or Pine Grove. The rain had the following median and range values: pH = 4.7 (4.1-6.2), SC = 18 μS/cm (6-78 μS/cm), sulfate = 2.4 mg/L (<1-5.9 mg/L), and total iron = 0.053 mg/L (0.043-0.077 mg/L). The pH, SC, and solute concentrations for the precipitation were consistently lower than those for Swatara Creek. Only one rain sample had pH > 5; the high pH value for this sample may represent the influence of dust or other debris.

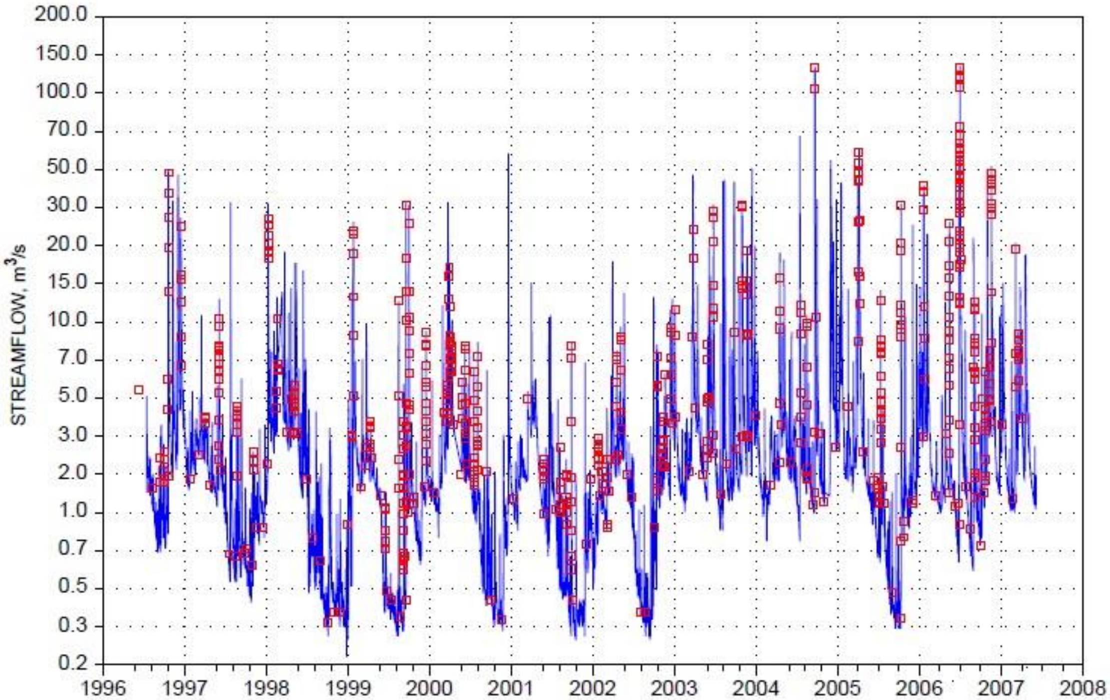

Figure 17. Streamflow hydrograph (recorded at 15-minute intervals) for Swatara Creek at Ravine, Pa., June 1996 through June 2007. Square symbols indicate instantaneous streamflow when water-quality samples were collected.

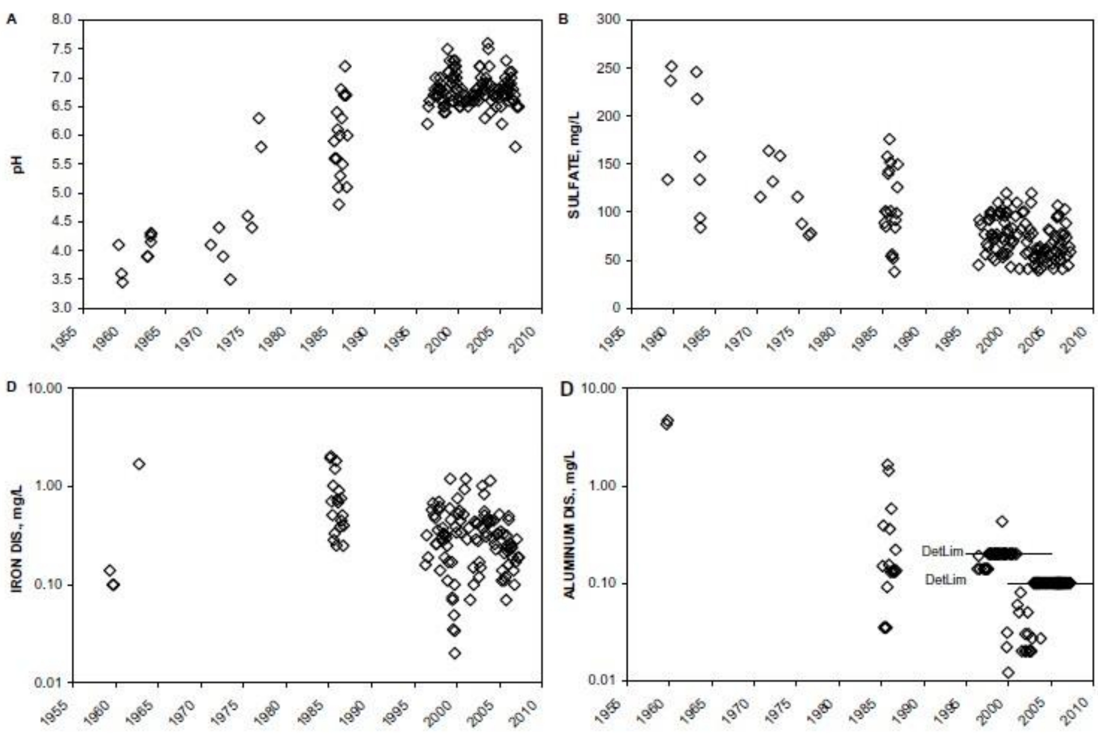

Figure 18. Long-term water-quality data for base flow of Swatara Creek at Ravine, 1959 - 2007: A, pH; B, dissolved sulfate; C, dissolved iron; D, dissolved aluminum. Data from McCarren et al. (1964), Stuart et al. (1967), Skelly & Loy, Inc. (1987), Fishel (1988), and U.S. Geological Survey (variously dated). Aluminum concentrations below detection are shown at detection limit (DetLim), indicated by horizontal line segments.

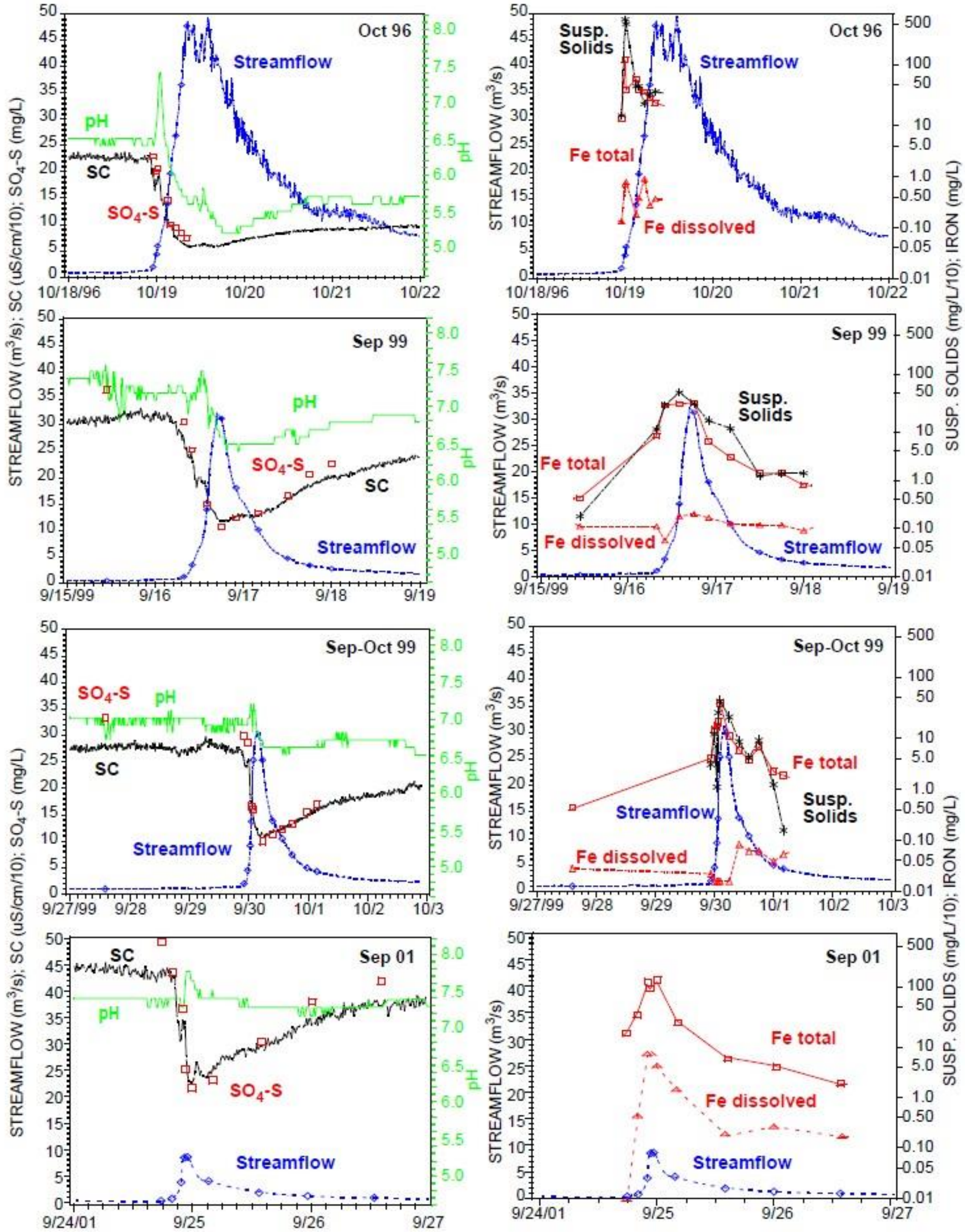

Figure 19. Hydrographs and associated water-quality data for selected stormflow events, Swatara Creek at Ravine, Pa. October 19-21, 1996; September 16-17, 1999; September 30-October 1, 1999; September 24-26, 2001; October 27-29, 2003; September 18-21, 2004; October 7-9, 2005; and September 2-4, 2006. Values shown for SC and suspended solids (divided by 10) and concentration of SO_4 (divided by 3) as sulfur.

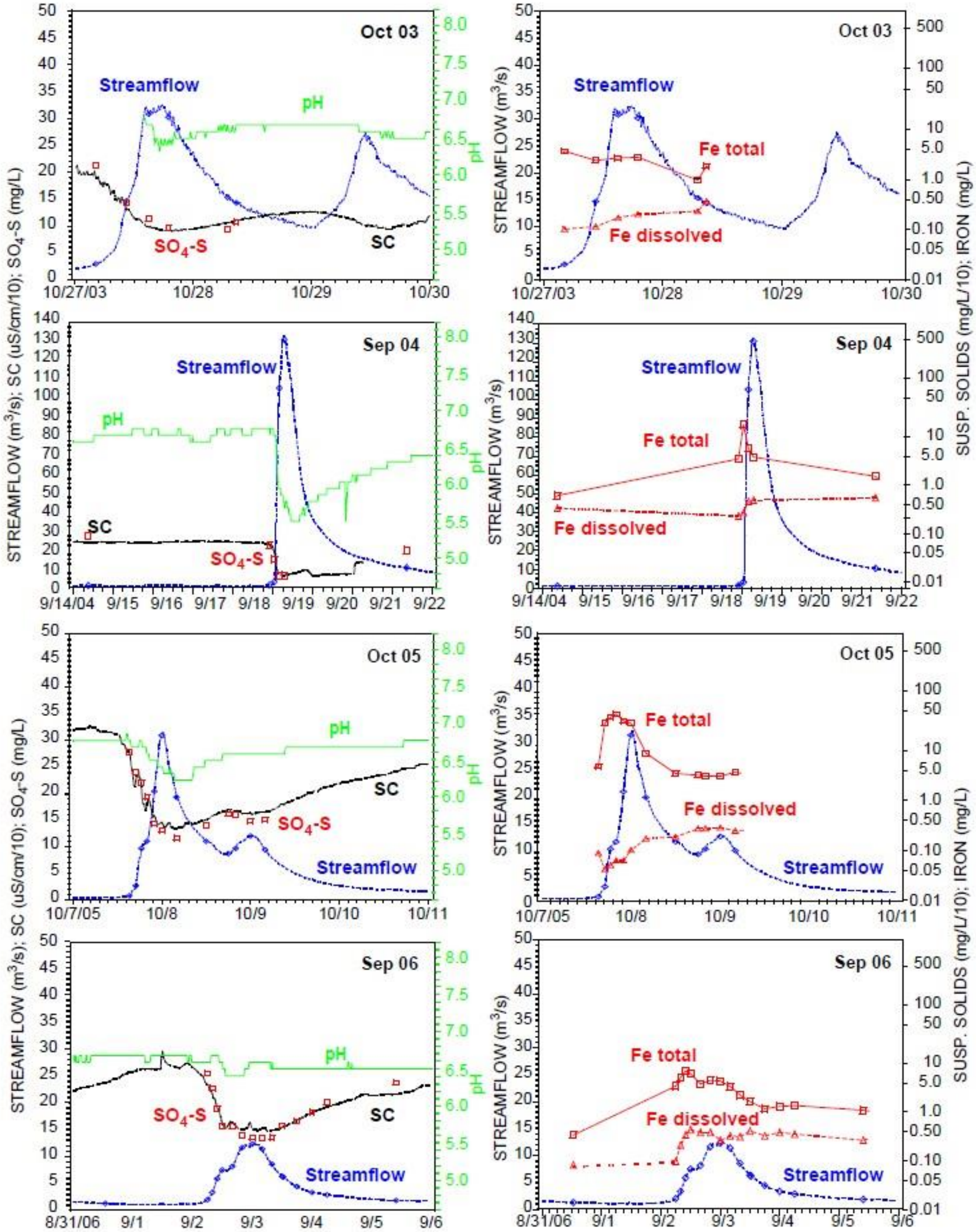

Figure 19. Cont.

<u>Return of Fish Populations, 1996-2006</u>

During the 1990s, native fish populations returned to upper Swatara Creek. No fish were found during ecological surveys of Swatara Creek at Ravine prior to 1990 (Bradford and Sickles 1950; Jackson 1987; Shoemaker 1932; Skelly & Loy, Inc. 1987). Yet, in 1996, six species of fish, including blacknose dace (*Rhinichthys atratulus*), brook trout (*Salvelinus fontinalis*), and white sucker (*Catostomus commersoni*) were captured by electrofishing (Table 6). The colonizing fish are assumed to have originated from wild stocks in unaffected or marginally affected tributaries and downstream reaches in the watershed. From 1996 to 2002, the number of fish species in Swatara Creek at Ravine increased annually to 25 species (Table 6, Figure 20). However, during high base-flow conditions in 2003 and 2004, fewer fish were captured than preceding years (Figure 20). When the surveys were resumed in 2005 and 2006, base-flow conditions were comparable to earlier survey conditions and large numbers of fish of various species were captured.

The number of fish species and total number of fish counted in Swatara Creek at Ravine were inversely related to the streamflow on the date of survey and, to a lesser extent, the maximum streamflow during the week of the survey (Figure 20). High base-flow conditions on the date of the survey increased water depth, turbidity, and velocity of transport of stunned fish resulting in reduced capture efficiency. Fish species that were relatively abundant during the higher streamflow conditions, notably rock bass (*Ambloplites rupestris*), were concentrated near large rocks and boulders along the stream bank and were more easily captured than other fish species at higher flows.

In 1996 and 2006, streamflow conditions of Swatara Creek at Ravine during the dates and weeks of fish surveys were similar (Figure 20). Despite similar survey conditions and methods, only 76 fish of 6 species were collected in 1996 compared to a total of 195 fish of 16 species in 2006. A fraction of the fish species identified at Ravine was found at upstream sites on Good Spring Creek at Tremont, Lorberry Creek at Lorberry Junction, and Swatara Creek at Newtown during the study (Table 6). Comparing survey results for 1996 and 2006, increases in fish-species diversity also were apparent for Good Spring Creek at Tremont (5 species in 1996; 9 species in 2006) and Swatara Creek at Newtown (0 fish in 1996; 2 brook trout in 2006) (Figure 20).

As indicated by boxplots summarizing water-quality data for the sites where fish surveys were conducted, Swatara Creek at Ravine and Good Spring Creek at Tremont generally exhibited net-alkaline water quality with consistently near-neutral pH during the study (Figure 21); these two sites also had the largest streamflow and yielded the greatest numbers of fish compared to Lorberry Creek and Swatara Creek at Newtown (Figure 20). In contrast, the water quality for Lorberry Creek and Swatara Creek at Newtown frequently was acidic with corresponding values of pH ranging to 5.5 and less during the study (Figure 21).

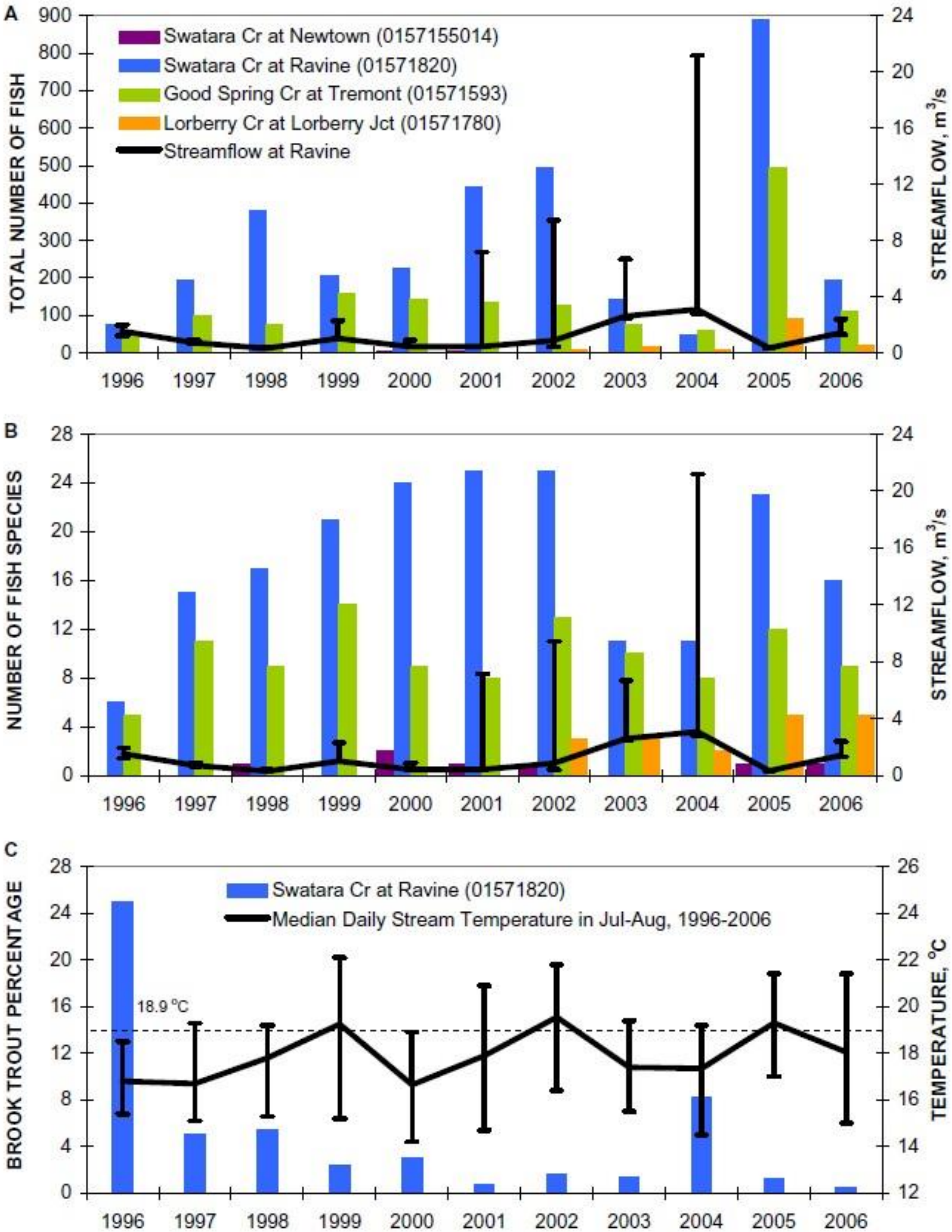

Figure 20. Annual electrofishing survey results at selected sites* in Swatara Creek Basin, 1996-2006: A, total number of fish at each site; B, number of fish species at each site; C, percentage of brook trout relative to total number of fish at Swatara Creek at Ravine. *Lorberry Creek was not surveyed before 2002. In A and B, solid black line indicates observed streamflow for Swatara Creek at Ravine during survey; vertical error bars indicate range of daily mean streamflow at Ravine during the week before the survey. In C, solid black line indicates daily mean temperature in July and August; vertical error bars indicate associated range of daily mean temperature; dashed horizontal line indicates maximum temperature permitted for "cold-water fishery" in July and August (Commonwealth of Pennsylvania 2002).

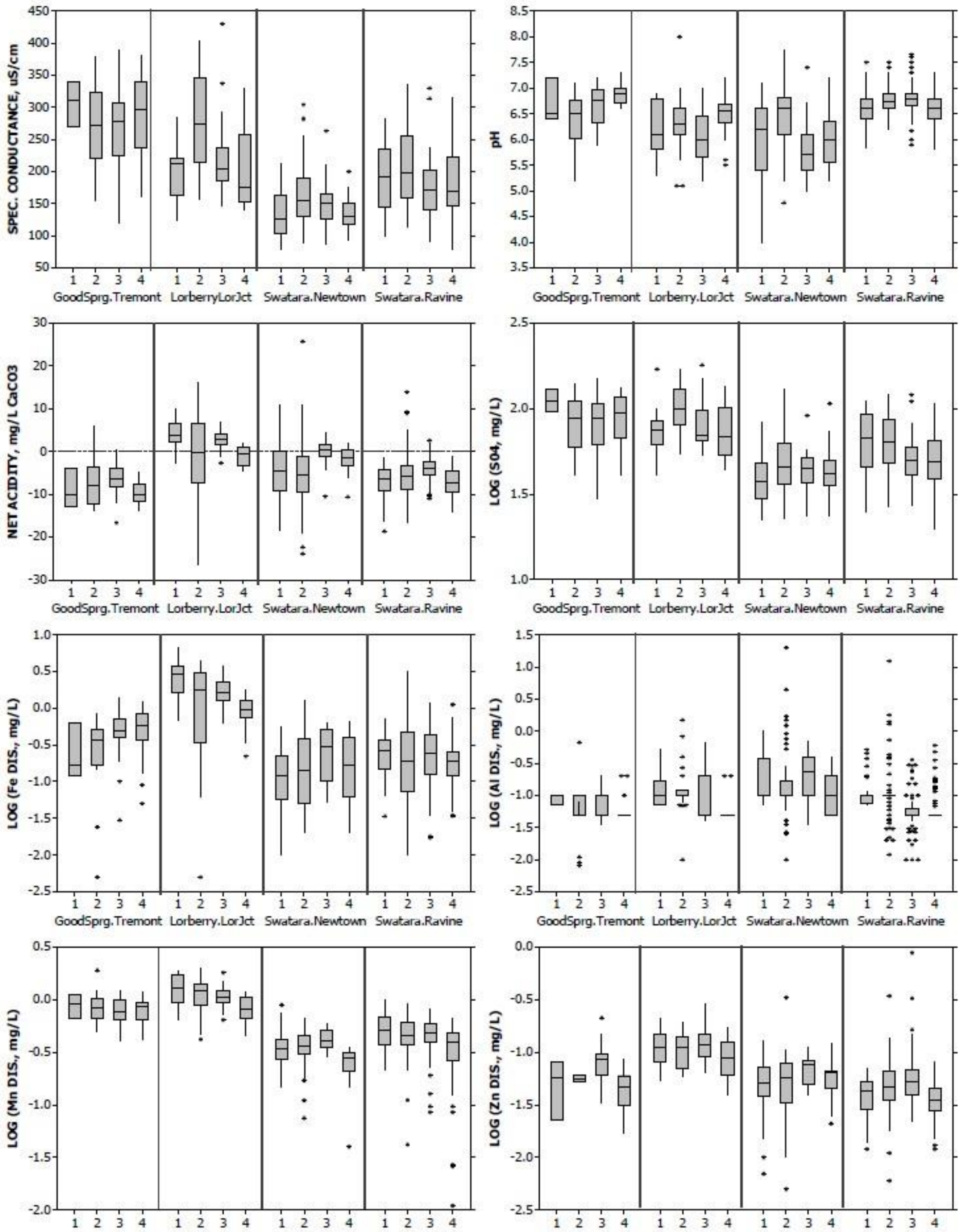

Figure 21. Boxplots summarizing hydrochemical characteristics of stream water at sites of annual fish surveys in Swatara Creek Basin, Pa., over 3-year intervals: (1) 1996-1998, (2) 1999-2001, (3) 2002- 2004, (4) 2005-2007. Shaded area of box indicates the "interquartile" range (IQR = 25th to 75th percentile); horizontal line inside the box indicates the median; vertical lines extend to extreme values within 1.5 times the IQR; symbols indicate outlier values.

Table 6. Fish species identified during annual surveys of Swatara Creek at Ravine, Pa., 1996-2006

Taxa[17]: ORDER / Family / *Genus species*	Common Name	Minimum pH in PA[18]	Pollution Tolerance[19]	7/96	10/97	9/98	9/99	10/00	10/01	10/02	10/03	10/04	10/05	10/06	Other Records[20]
				Number of Individuals (Month and Year of Survey)											
CYPRINIFORMES															
Cyprinidae															
Campostoma	Central stoneroller	6.0	M	0	0	0	2	35	67	69	5	1	23	6	LG
Cyprinella spiloptera	Spotfin shiner	6.4	M	0	0	3	3	0	1	0	0	0	0	0	
Exoglossum	Cutlips minnow	6.1	I	0	1	0	0	1	1	1	0	0	0	0	G
Luxilus cornutus	Common shiner	6.0	M	0	0	0	0	0	6	1	0	0	1	0	G
Nocomis micropogon	River chub	6.0	I	1	14	9	44	27	75	76	7	2	26	9	G
Notemigonus	Golden shiner	4.6	T	0	0	1	0	1	0	0	0	0	0	0	
Notropis amoenus	Comely shiner	6.5	T	0	0	0	0	0	1	0	0	0	0	0	G
Notropis hudsonius	Spottail shiner	6.4	M	0	0	0	0	2	0	0	0	1	0	0	
Notropis rubellus	Rosyface shiner	6.0	I	0	1	0	0	0	0	0	0	0	1	2	
Pimephales notatus	Bluntnose minnow	5.6	T	0	0	0	0	0	0	0	0	0	1	1	G
Rhinichthys atratulus	Blacknose dace	5.6	T	22	47	16	6	46	26	99	6	2	10	22	NLG
Rhinichthys cataractae	Longnose dace	5.9	I	12	1	17	4	24	28	15	0	0	10	0	LG
Semotilus	Creek chub	5.2	T	0	7	22	1	7	2	32	0	0	8	2	NLG
Semolitus corporalis	Fallfish	6.1	M	0	66	54	30	20	12	49	1	1	23	22	G
Catostomidae															
Catostomus	White sucker	4.6	T	20	25	52	22	19	35	26	2	1	43	6	G
Hypentelium nigricans	Northern hog sucker	6.0	I	0	0	0	5	2	0	1	0	3	30	3	
SILURIFORMES															
Ictaluridae															
Ameiurus natalis	Yellow bullhead	6.5	T	0	1	1	0	1	2	1	2	0	2	0	
Ameiurus nebulosus	Brown bullhead	4.6	T	0	1	12	2	1	0	1	0	0	1	2	G
Noturus insignis	Margined madtom	5.9	M	0	0	2	9	3	9	2	1	0	6	0	G
ESOCIFORMES															
Esocidae															
Esox niger	Chain pickerel	4.6	M	0	0	0	2	0	1	4	0	0	0	0	G
SALMONIFORMES															
Salmonidae															
Oncorhynchus mykiss	Rainbow trout	6.5	M	0	0	0	0	0	1	0	0	0	0	0	G
Salmo trutta	Brown trout	5.9	M	2	0	1	2	1	2	3	0	0	1	0	LG
Salvelinus fontinalis	Brook trout	5.0	M	19	10	21	5	7	3	8	2	4	11	1	NLG
SCORPAENIFORMES															
Cottidae															
Cottus sp.	Sculpin	5.9	M	0	0	2	0	2	1	3	0	0	1	1	
PERCIFORMES															
Centrarchidae															
Ambloplites rupestris	Rock bass	6.0	M	0	0	0	6	5	20	66	10	31	15	11	G
Lepomis auritus	Redbreast sunfish	6.2	M	0	0	2	2	4	12	1	0	0	0	0	G
Lepomis cyanellus	Green sunfish	6.4	T	0	0	0	0	0	2	2	0	0	5	0	
Lepomis gibbosus	Pumpkinseed	4.6	M	0	1	0	2	3	7	3	0	0	0	0	G
Lepomis macrochirus	Bluegill	6.5	M	0	0	2	1	1	0	6	0	1	0	2	G
Micropterus dolomieu	Smallmouth bass	6.0	M	0	7	0	52	4	12	12	5	0	17	3	G
Micropterus salmoides	Largemouth bass	4.7	M	0	1	0	0	0	0	0	0	0	2	0	
Percidae															
Etheostoma olmstedi	Tessellated darter	5.9	M	0	12	16	3	6	5	8	2	0	27	3	

17 Names are consistent with the Pennsylvania species taxa list of Steiner (2000).
18 Minimum pH of occurrence in freshwater in Pennsylvania as reported by Butler et al. (1973).
19 Pollution tolerance: I (intolerant), M (moderate), T (tolerant), adapted from Barbour et al. (1999)
20 Letter indicates fish species identified during 1996-2006 at other stations: N, Swatara Creek at Newtown; L, Lorberry Creek at Lorberry Junction; G, Good Spring Creek at Tremont.

Percina peltata	Shield darter	6.5	I	0	0	0	3	2	3	6	0	1	13	0
Total number of individuals collected:				76	19	37	20	22	44	49	14	48	89	19
Total number of species identified:				6	15	17	21	24	25	25	11	11	23	16

Macroinvertebrates

Results of the macroinvertebrate sampling were shown in a paper presented by Cravotta and Bilger (2001). In general the benthic community at Ravine has not exhibited the same dynamic as the fish community. However, an improved water-quality trend from 1994 to 1999 is implied by an increased abundance of taxa that are considered intolerant of pollution (Table 7). In 1994 and 1996, six taxa (family level) were recorded; during 1997-1999, 8 to 11 taxa were recorded. The calculated Hilsenhoff's (1988) family-level biotic index (Table 7) indicates water quality at Ravine improved, from fair in 1994 to very good in 1999. Hydropsychidae (caddisflies) and Chironomidae (midges), which are known to tolerate acidic conditions, were consistently dominant during 1994-1999 (Table 7). Although not as many, the appearance of Ephmeroptera (mayflies), including Baetidae and Heptageniidae, in 1997 and later years is significant in that these insects are sensitive to acidic conditions and considered intolerant to pollution (Table 7).

Data was not collected during 2001-2006. In September 2007, a macroinvertebrate study was completed by Department of Environmental Protection with result very similar of past studies (Gary Walters, Pennsylvania Department of Environmental Protection, 2008, written communication).

Table 7 Benthic macroinvertebrates identified in Swatara Creek at Ravine, Pa., 1994-1999[21]

Taxa ORDER Family *Genus*	Pollution Tolerance[22]	Month and Year of Survey					
		8/94	10/96	9/97	9/98	9/99	9/99
		Number of Individuals					
EPHEMEROPTERA (mayflies)							
Baetidae	4						
Acentrella		0	0	3	2	1	2
Baetis		0	0	14	22	4	5
Heptageniidae	4						1
Stenacron		0	0	0	0	1	0
Stenonema		0	0	1	0	1	0
Plecoptera (stoneflies)							
Leuctridae	0						
Leuctra		1	1	0	2	0	0
MEGALOPTERA (dobsonflies, alderflies)							
Sialidae	4						
Sialis		1	3	0	2	1	1
COLEOPTERA (aquatic beetles)							
Dryopidae	5						
Helichus		0	0	1	0	0	0
Elmidae	4						
Optioservus		0	0	0	1	0	0
Promoresia		0	0	0	0	1	0
Stenelmis		0	0	1	0	0	1
Psephenidae	4						
Psephenus		0	0	0	0	1	0
TricHoptera (caddisflies)							
Hydropsychidae	4						
Ceratopsyche		0	0	0	0	0	33
Cheumatopsyche		0	0	0	14	12	8
Diplectrona		2	0	5	1	0	0
Hydropsyche		18	12	25	59	40	39
Philopotamidae	3						
Dolophilodes		0	0	0	0	0	1
Rhyacophilidae	0						
Rhyacophila		0	0	1	0	0	2
Diptera (true flies)							
Chironomidae	6	33	0	63	21	12	6
Empididae	6						
Chelifera		1	5	1	11	0	0
Hemerodromia		2	6	0	0	0	2
Tipulidae	3						
Antocha		0	0	0	1	0	0
Dicranota		0	1	0	3	5	4
Limnophila		0	0	0	0	0	1
NON-INSECT TAXA							
DECAPODA (crayfish)							
Cambaridae	6						

21 Taxa identified in 1994-99 by D. Bogar, PaDEP, and independently in 1999 by M. D. Bilger, USGS (last column).

22 Pollution tolerance index values from 0 to 10 and number of individuals used to compute Hilsenhoff's (1988) family-level biotic index: 0.00-3.75 (excellent), 3.76-4.25 (very good), 4.26-5.00 (good), 5.01-5.75 (fair), 5.76-6.50 (fairly poor), 6.51-7.25 (poor), and 7.26-10.00 (very poor) (U.S. Environmental Protection Agency 1993).

Cambarus		0	1	0	0	0	0
HYDRACHNIDIA (water mites)	4	0	0	0	0	0	2
OLIGOCHAETA (aquatic earthworms)							
Lumbricidae	6	3	0	2	0	0	0
Total number of individuals collected:		**61**	**29**	**117**	**139**	**79**	**108**
Total number of taxa identified (family level):		**6**	**6**	**9**	**8**	**8**	**11**
Hilsenhoff's family-level biotic index[2]		**5.21**	**4.65**	**5.10**	**4.37**	**4.24**	**4.02**

Discussion

Discussion of Treatment Performance

Results of monitoring during 1996-2007 of six limestone treatment systems designed for acid removal and two wetland systems designed to remove precipitated metals indicate the anoxic limestone drain on the Buck Mountain discharge (Site A1 in Figure 2) near the headwaters of Swatara Creek had the greatest overall benefit. This anoxic limestone drain, which had been in service for more than 10 years, consistently exported an annual load of $CaCO_3$ greater than 26 t/y, equivalent to the acid removed, and produced significant improvement in pH of downstream water for relatively low cost. Compared to the 29 anoxic limestone drains evaluated by Ziemkiewicz et al. (2003), the median flow rate treated by the Buck Mountain anoxic limestone drain was two times greater than the highest they reported, and the acid-removal efficiency was near the median value for other anoxic limestone drains. However, the cost efficiency for the Buck Mountain anoxic limestone drain was greater than 75 percent of the anoxic limestone drains evaluated by Ziemkiewicz et al. (2003), reflecting added expenses incurred for twice enlarging the Buck Mountain treatment system in 2001 and 2005.

The other treatment systems in the upper Swatara Creek Basin had treatment efficiencies and cost efficiencies within the ranges reported by Ziemkiewicz et al. (2003). The limestone-sand treatment on Coal Run (LSC) was relatively effective and the least expensive for acid removal. The open limestone channel on Swatara Creek (OLS) was among the most expensive per ton of acid removed. The oxic limestone drain on the Hegins discharge (ODH) and the limestone diversion wells on Swatara Creek (DWS) and Lorberry Creek (DWL) were intermediate in treatment and cost efficiency.

On average, the diversion wells on Lorberry Creek and Swatara Creek treated a larger flow volume than the other treatment systems. The diversion wells were effective in removing acidity and increasing pH of downstream water and exhibited unique potential to treat moderate to high flows. Because stormflow generally was more acidic than base flow in the Swatara Creek, diversion wells could be useful to augment treatments by other limestone-based systems at upstream or downstream sites. However, diversion-well systems are relatively expensive to operate because they require routine maintenance to ensure that they contain sufficient limestone through the duration of a treatment event and that they do not become clogged with debris. Although a large fraction of the streamflow bypassed the diversion wells on Swatara Creek and Lorberry Creek during the highest flow conditions, multiple diversion wells with intakes at higher elevations than normal base-flow stage could be added to treat progressively larger volumes during such stormflow events.

At near-neutral pH, the transport of dissolved Fe, Mn, and Al in AMD can be attenuated by precipitation of oxyhydroxides. However, the precipitation of Fe and Mn oxyhydroxides requires oxidation of the dissolved metals. Although associated trace metals, including Ni and Zn, tend to adsorb on Fe^{III}, $Mn^{III\text{-}IV}$, and Al oxyhydroxides at near-neutral pH, slow rates of oxidation limit passive treatment and metal-removal efficiency (e.g. Watzlaf et al. 2004; Cravotta 2007). Wetlands installed along Lorberry Creek (WLL) and on Lower Rausch Creek (WLR) were effective at reducing metals transport to downstream sites because they increased the time available (retention time) for Fe and Mn oxidation and provided a location for removal of the metal-rich solids. In addition, the limestone-compost substrate of the Lower Rausch Creek wetlands apparently provided for sulfate reduction and associated alkalinity production.

Nevertheless, both of the wetland treatment systems promoted increases in water temperature during summer months that could have adverse effects on fish in downstream reaches.

Although the study spanned more than 10 years, extended monitoring and documentation of treatment-system maintenance in the Swatara Creek Basin could be helpful to indicate long-term performance of the treatment systems as they approach the end of their service life. The cost efficiency computed by Ziemkiewicz et al. (2003) and in this paper assumed a 20-year service life for all treatment systems and implied that treatment performance (e.g. acid removal) would be maintained for the duration. Consideration of a future service life is useful for normalizing performance results; however, the assumed 20-year service life may be unrealistic. Furthermore, declines in performance can be expected as the treatment substrate is consumed or retention time is reduced (e.g. Cravotta 2003, 2008b). Specifically, several treatment systems evaluated in this paper required major maintenance or reconstruction within 10 years of implementation. Although limestone drains may be considered “passive” treatment systems, which involve minimal maintenance, the anoxic limestone drain at the Buck Mountain discharge and the oxic limestone drains at the Orchard and Hegins discharges all required replenishment of limestone to ensure continued benefits. Furthermore, periodic flushing of precipitated solids from the limestone beds may be necessary. Because of the high level of maintenance, diversion wells are classified as a “semi-passive” treatment (Skousen et al. 1998). As designed, the limestone diversion wells required frequent additions of limestone and occasional clearing of pipes. Other treatments such as the limestone channel and limestone-sand dosing could require periodic replenishment of limestone, plus the wetlands could require sludge removal to maintain performance results.

Correlations among Streamflow, Metals, and Suspended Solids

The pH, SC, sulfate, and other chemical concentrations varied in response to changes in streamflow. Generally, base-flow samples had higher pH, SC, alkalinity, hardness, and concentrations of dissolved major ions and lower concentrations of total metals compared to stormflow (Figures 19 and 22).

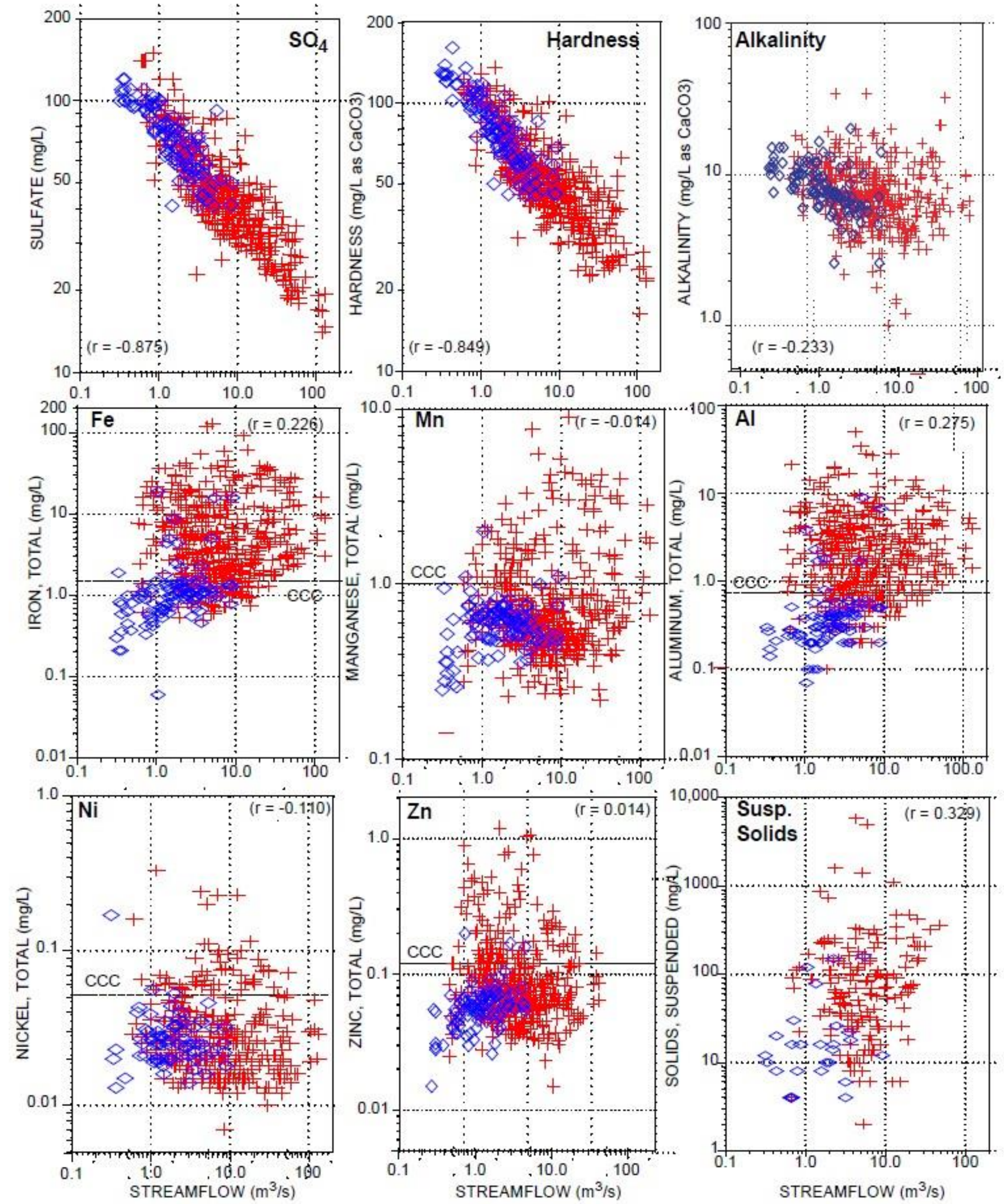

Figure 22. Relations between streamflow and concentrations of water-quality constituents in base flow (open diamond symbol) and stormflow (cross symbol) samples, Swatara Creek at Ravine, Pa. Hardness was computed from dissolved Ca and Mg in milligrams per liter ($2.5.C_{Ca}$ + $4.1.C_{Mg}$). Spearman rank correlation coefficient, r; values > 0.138 or < -0.138 are significant ($p<0.001$). Dashed horizontal lines, except for Mn, indicate criteria continuous concentration (CCC) values for protection of freshwater aquatic organisms (U.S. Environmental Protection Agency, 2002); dashed lines for Mn indicate PaDEP standard for daily mean concentration (Commonwealth of Pennsylvania 2002).

As streamflow at Ravine increased during stormflow events, pH, SC, and concentrations of sulfate and manganese typically decreased, and concentrations of suspended solids, iron, aluminum, and other metals in whole-water samples typically increased (Figures 19 and 22). Similar trends for dissolved and suspended solids during stormflow on Swatara Creek in 1959 were reported by Stuart et al. (1967, Figure 19). However, the trends for pH, SC, and sulfate are inconsistent with the work of others who evaluated impacts of acid rain on small streams in unmined, forested watersheds of the Appalachian Mountains of northeastern USA. For example, Corbett and Lynch (1982) and DeWalle (1990) showed pH typically *decreased* while sulfate *increased* with streamflow in Appalachian headwater streams during storm events.

Cravotta (2000) demonstrated that the decreases in pH, SC, and concentrations of major ions during storm events for Swatara Creek could result from mixing of weakly acidic storm runoff having pH 4.0-4.5 and low dissolved solids with poorly buffered stream water having pH 6.0-6.5 and high sulfate. The storm runoff is derived from acidic rainfall with minor contributions from pyrite-oxidation products and carbonate minerals (e.g. Olyphant et al. 1991).

Typically, the greatest changes in SC and pH occurred with the largest changes in streamflow (greatest dilution by storm runoff). The minimum SC typically occurred with peak streamflow, whereas the minimum pH lagged by several hours, generally occurring during the falling stage. In contrast, concentrations of suspended solids generally increased to peak values during the initial rising stage and decreased prior to peak stage. Although the concentration of total iron included contributions from suspended particles, peaks for total iron tended to be achieved after the peaks for suspended solids, possibly reflecting a time lag for iron-laden water and associated sediment from the upper, mined part of the watershed to reach Ravine. Generally, concentrations of suspended solids and total iron and other metals at a given streamflow during a storm event were greater during the rising stage than the falling stage (Figure 19). This "hysteresis" effect can be explained as resulting from the accumulation of metal-rich sediments (Fe^{III}, Mn^{III-IV}, and Al oxyhydroxides and clay minerals) within the streambed during base-flow conditions, scour and transport of the streambed deposits during rising stormflow stage, and dilution during falling stages. Small storm events can scour metal-rich sediments from the streambed with little dilution of the concentrations, resulting in concentrations of total metals and suspended solids that are comparable with or greater than those of large storms. Stormflow hysteresis patterns indicated for Swatara Creek and other streams can be affected by preceding conditions, with large peak concentrations following relatively stable base flow and diminished peak concentrations during succeeding storms of the same magnitude (Bowes et al. 2005; Caruso et al. 2008).

Because of the hysteresis effect, streamflow and concentrations of metals in Swatara Creek at Ravine were poorly correlated (iron and aluminum) or not correlated (manganese, nickel, zinc) (Figure 23). However, concentrations of total metals were strongly correlated with the concentration of suspended solids (Cravotta and Bilger 2001). The correlations between concentrations of suspended solids and total metals are consistent with suspended solids that contained approximately 10 % iron, 5 % aluminum, and lesser amounts of manganese and trace metals, which were the reported concentrations in fine streambed sediments in the study area (Cravotta and Bilger 2001).

Concentrations of aluminum, nickel, zinc, and other trace metals commonly were detected in the unfiltered samples but not in the corresponding filtered samples. Hence, the "dissolved" chemical concentrations did not include substantial contributions from <0.45-μm colloids (e.g. Kimball et al. 1995; Schemel et al. 2000). Furthermore, when detectable in both unfiltered and filtered samples, the total concentrations of iron and aluminum, and, to a lesser extent, manganese, nickel, and zinc commonly exceeded those in filtered samples (Figure 23) indicating a major fraction of these metals was associated with suspended particles. In contrast, equivalent values for total and dissolved concentrations of manganese, nickel, and zinc frequently were reported in base-flow samples (Figure 23), indicating a major fraction of these metals was transported as dissolved ions and, possibly, fine colloids that could pass through filters.

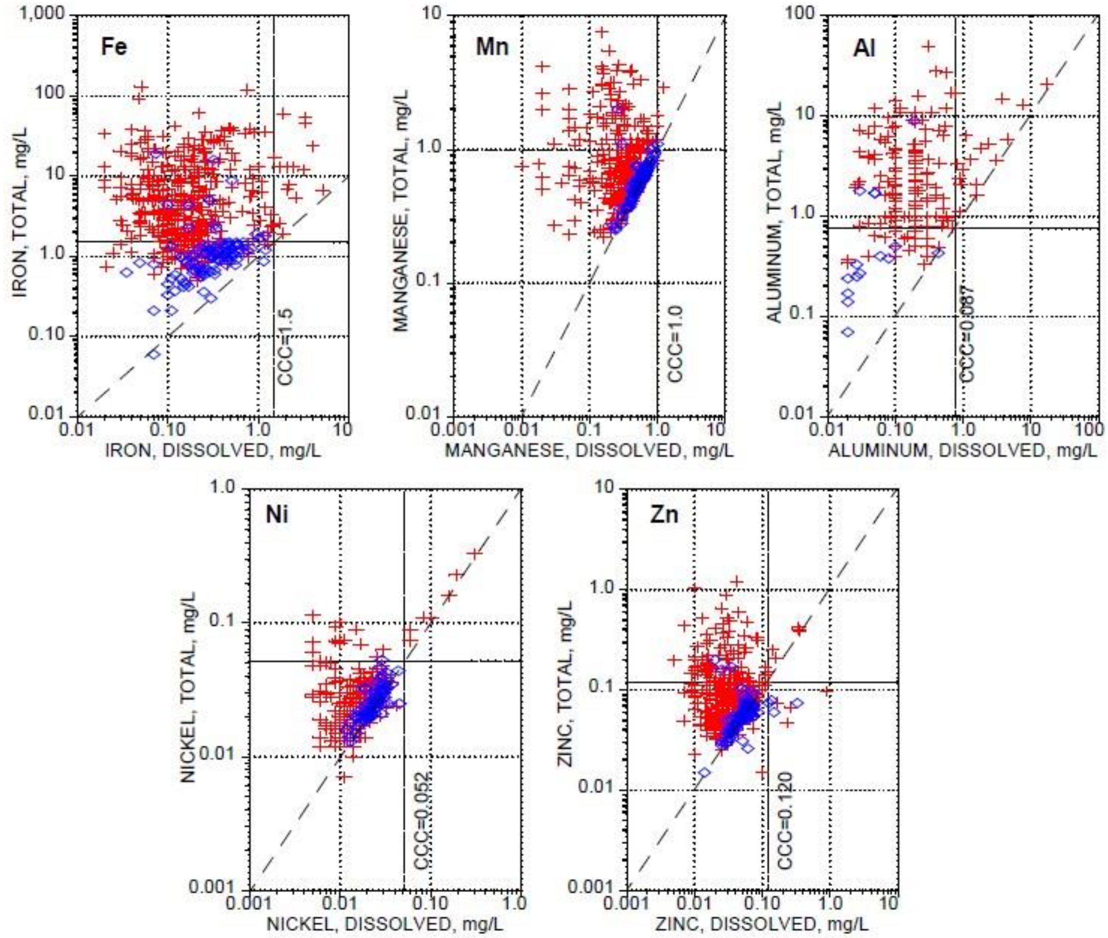

Figure 23. Relations among concentrations of dissolved and total metals in stream water sampled during base-flow and stormflow conditions, Swatara Creek at Ravine, Pa. Values farther to right of diagonal line indicate decreasing fraction of dissolved ions (<0.45 μm) contributing to total concentration. Data plotted only if total and dissolved concentration above limit of detection. Dotted horizontal and vertical lines, except for manganese, indicate criteria continuous concentration (CCC) values for protection of freshwater aquatic organisms (U.S. Environmental Protection Agency 2002); dotted lines for manganese indicate PaDEP standard for daily mean concentration (Commonwealth of Pennsylvania 2002).

<u>Water-Quality Trends</u>

Continuous-record data for streamflow, pH, SC, and temperature for Swatara Creek at Ravine during the entire study could be evaluated directly to indicate temporal differences (e.g. Figures 14 and 15). However, the interpretation of trends in concentrations and loads of chemicals collected at different time intervals was complicated by the effects of changing streamflow on the pH and chemical concentrations (Figures 16, 17, and 22). Thus, continuous streamflow data were used with Eq. (1) to estimate daily loads and annual flow- weighted concentration (FWC) values for the study period. The use of these estimates could help to remove sampling bias and facilitate the interpretation of water-quality trends that resulted from factors other than changes in streamflow.

As expected because of autocorrelation, the annual streamflow and annual loads for all chemicals changed in parallel (Figure 24). However, the FWC values for different chemicals exhibited temporal variations not correlated with streamflow (Figure 24). For Swatara Creek at Ravine, the FWC values for hydrogen ion, alkalinity, and dissolved iron had similar trends, decreasing from high values during 1997-1998 to minimum values in 2001-2003 and then increasing during 2003-2006. In contrast, FWC values for manganese and, to a lesser extent, sulfate exhibited possible downward trends, whereas those for dissolved aluminum were more erratic.

For Swatara Creek at Newtown, FWC estimates were computed for the sites upstream (0157155010) and downstream (0157155014) of limestone diversion wells using the streamflow record from the downstream site (Figure 25). During 1997-2003, the FWC values for hydrogen ion and metals were lower and those for alkalinity were higher at the downstream site compared to the upstream site. These differences in water quality between the two sites were expected because of the continuous addition of alkalinity and pulverized limestone to the stream by the diversion wells. However, the diversion wells were damaged by storms associated with Hurricane Ivan in September 2004 and were not operated continuously thereafter. After 2004, the FWC values for hydrogen ion increased and those for alkalinity decreased at the downstream site, while differences between the FWC values at the two sites became smaller for dissolved iron and manganese.

Flow-adjusted trends, which are identical for concentration and load of the particular chemical, were expressed as percent change between the 1997 start time and 2006 end time (period of continuous streamflow record). Flow-adjusted trends for Swatara Creek at Ravine (Figure 26) indicated significant decreases in hydrogen ion, dissolved and total manganese, total iron, and dissolved aluminum; no change in alkalinity, sulfate, or dissolved iron; and increases in calcium. The lack of trend in sulfate could indicate that the AMD contaminant loading rate was unchanged during the study. The decrease in hydrogen ion and increase in calcium could result from the dissolution of limestone in various AMD treatment systems. Although generated by limestone dissolution, the lack of trend in alkalinity could indicate alkalinity was consumed during neutralization reactions that buffered the pH to be near neutral. Combined with decreases in iron, manganese, and aluminum, these flow-adjusted trends support the hypothesis that AMD treatment has increased pH and decreased the transport of dissolved metals during the study.

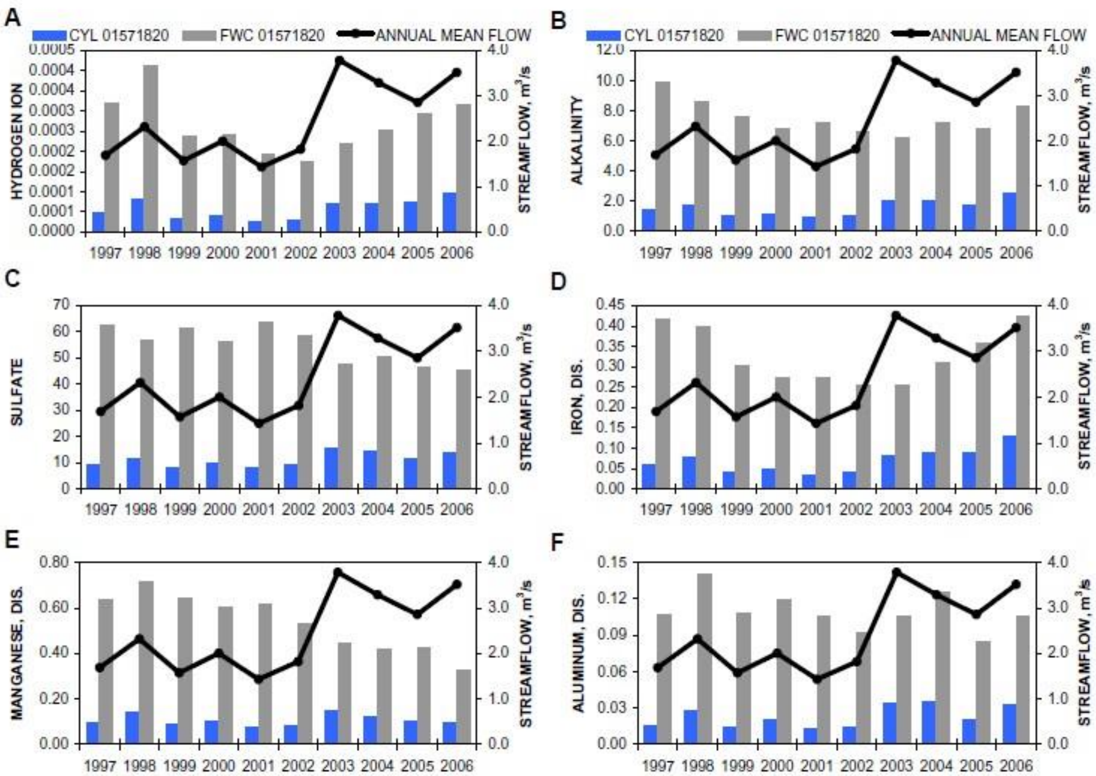

Figure 24. Annual mean streamflow for Swatara Creek at Ravine (01571820; black line) and corresponding loading by calendar year (CYL; left bar, units Mg/day) and flow-weighted concentration (FWC; right bar, units mg/L) of chemicals associated with mine effluent, 1997-2006: A, hydrogen ion; B, alkalinity; C, sulfate; D, dissolved iron; E, dissolved manganese; F, dissolved aluminum.

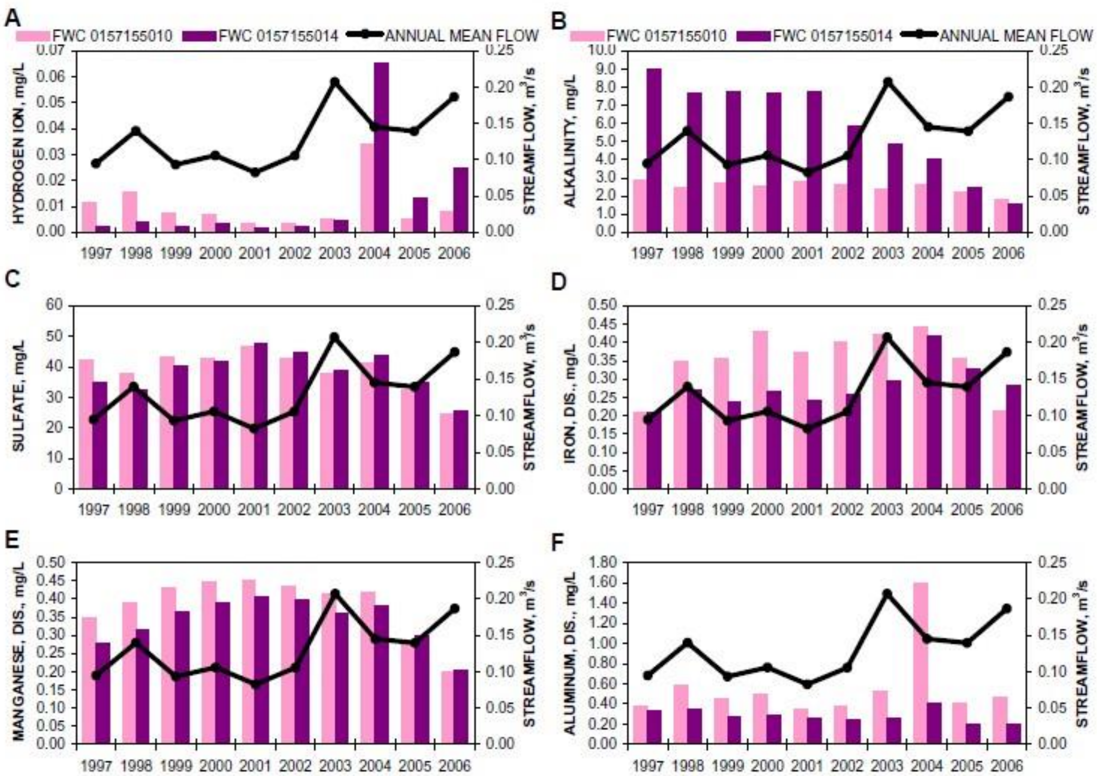

Figure 25. Annual mean streamflow for Swatara Creek at Newtown (0157155014; black bar) and corresponding flow-weighted concentration (FWC) of chemicals upstream (0157155010; left bar) and downstream (0157155014; right bar) of diversion wells, 1997-2006: A, hydrogen ion; B, alkalinity; C, sulfate; D, dissolved iron; E, dissolved manganese; F, dissolved aluminum.

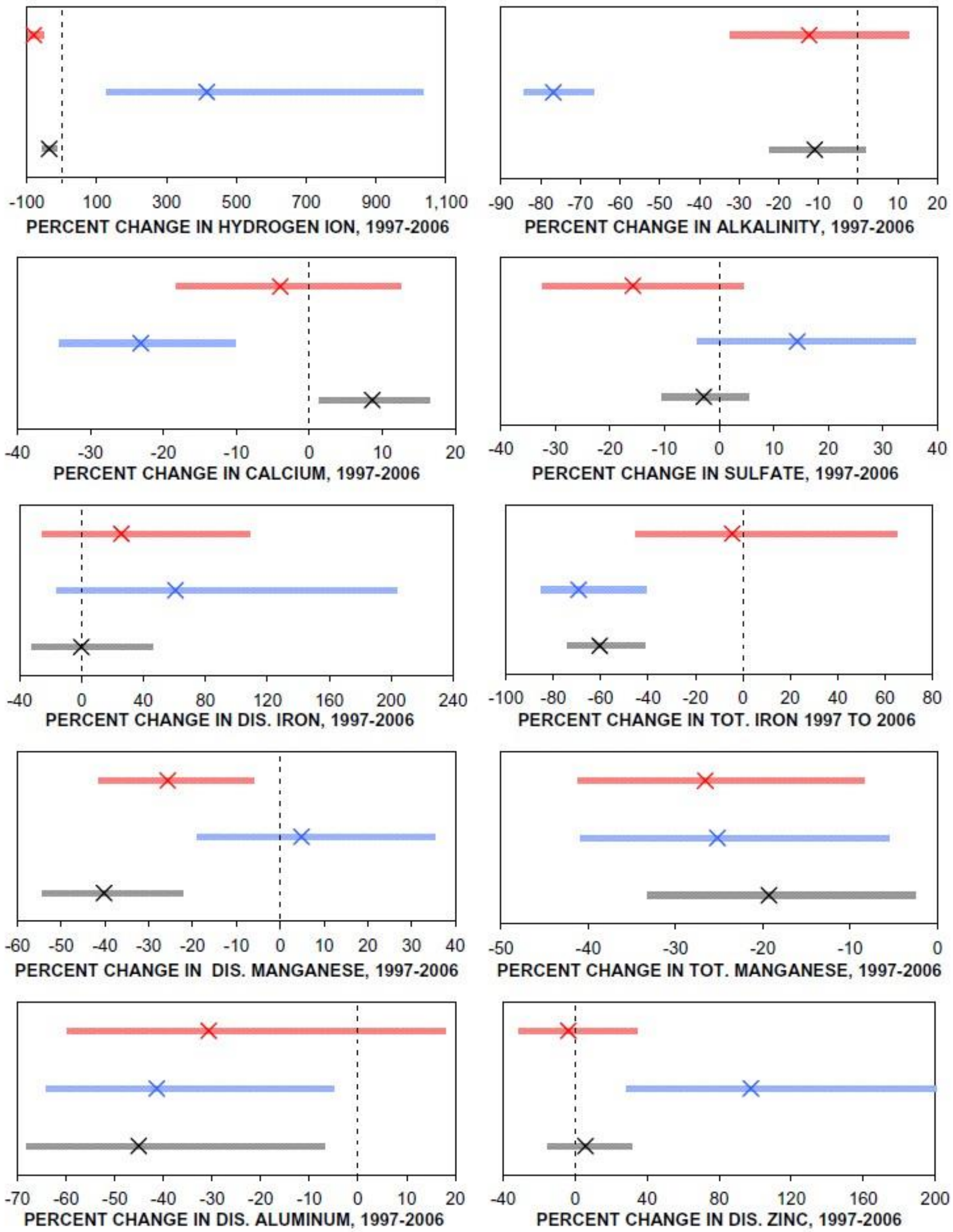

Figure 26. Estimated flow-adjusted trend (X) and confidence interval (CI) bar for chemicals in Swatara Creek at Ravine (01571820; lower black bar) and Swatara Creek at Newtown downstream (0157155014; middle blue bar) and upstream from diversion wells (0157155010; upper red bar), 1997-2006. If the CI is completely negative or completely positive, the trend is significant.

<u>Ecological Ramifications</u>

The increase in fish populations of Swatara Creek and its tributaries during the late 1990s coincided with the implementation of limestone-based treatment systems at many of the AMD sources (Figure 2). Possible effects of such treatments include increased concentrations of calcium and alkalinity with associated buffering of pH to be near neutral, which could benefit fish and other aquatic organisms that are intolerant of low pH and sensitive to toxic metals. Because of solubility and adsorption, the concentrations of dissolved metals would tend to decrease with increased pH (e.g. Cravotta 2008; Webster et al. 1998), plus added calcium may be important in regulating toxic effects of metals (Holt and Yan 2003; U.S. Environmental Protection Agency 2002; Yan et al. 2003). Flow-adjusted trends for Swatara Creek at Ravine indicating decreases in hydrogen ion and metals and increases in calcium during the 1997-2006 time period (Figure 26) and consistently near-neutral pH during 1999-2007 (Figures 14B, 15B, and 19) imply that the AMD treatments installed during 1995-2001 have helped to improve downstream water quality.

To maintain its designated use as a cold-water fishery, Swatara Creek and other such streams in Pennsylvania must have DO concentrations greater than 5.0 mg/L at all times and temperatures less than 18.9 °C during July and August (warmest months) (Commonwealth of Pennsylvania 2002). The minimum DO concentration at Ravine was 8.7 mg/L during July 1997. However, the streamwater temperature occasionally exceeded 18.9 °C during low-flow conditions in summer (Figures 20C and 15C), and concentrations of metals periodically exceeded water-quality criteria for protection of aquatic organisms (Figures 22 and 23). Although elevated temperatures can produce faster rates of iron oxidation and associated metals removal in AMD treatment systems (e.g. Cravotta 2007; Watzlaf et al. 2004), the prolonged exposure of stream water or AMD to ambient air temperatures or sunlight can produce temperature extremes that are not suitable for brook trout and other cold-water species.

The overall fish-community structure in Swatara Creek at Ravine could be characterized as transitional between cold-water and warm-water classifications. Although species abundance varied from year to year, the majority of the species collected during 1996-2006 was considered to have moderate tolerance to low pH and pollution (Table 6). Nevertheless, several of the fish taxa were intolerant of pollution and low pH, such as river chub (*Nocomis micropogon*), longnose dace (*Rhinichthys cataractae*), northern hog sucker (*Hypentelium nigricans*), and shield darter (*Percina peltata*) (Table 6). As the maximum stream temperature during summer months increased (Figures 15C and 20C), competition between cold-water and warm- water species could have been a factor affecting species abundance. For example, at Ravine during 1997- 1998, cold-water and cool-water species predominated, including blacknose dace, creek chub (*Semotilus atromaculatus*), fallfish (*Semolitus corporalis*), white sucker, brook trout, and tessellated darter (*Etheostoma olmstedi*) (Table 6). In 1999, cool-water species including smallmouth bass (*Micropterus dolomieu*), river chub (*Nocomis micropogon*), and fallfish were dominant, with substantially fewer blacknose dace, tessellated darter, and brook trout. Likewise, when rock bass, a warm-water species, were abundant in 2003 and 2006, numbers of brook trout were greatly diminished, possibly reflecting variations in streamflow during the survey in addition to the variations in maximum stream temperature (Figure 20). As observed elsewhere (Snucins and Gunn 2003), the range expansion of smallmouth bass and associated warm-water fish could be an important factor affecting food-web structure and the recovery of trout and associated cold-water fish in acid-stressed systems.

Base flow during the study met Commonwealth of Pennsylvania (2002) chemical water-quality standards; however, *stormflow* commonly *did not* meet standards for pH (6.0 to 9.0) or concentrations of total iron (1.5 mg/L daily mean), dissolved iron (0.3 mg/L maximum), and total manganese (1.0 mg/L maximum) (Figures 22 and 23). Furthermore, although concentrations of "dissolved" metals in filtered samples generally met U.S. Environmental Protection Agency (USEPA) criteria continuous concentration (CCC) limits for protection of freshwater aquatic organisms, the concentrations of "total recoverable" metals in unfiltered stormflow samples (Figures 22 and 23) commonly exceeded CCC values for iron (1.0 mg/L) and aluminum (0.087 mg/L) and occasionally exceeded CCC values for nickel (0.052 mg/L) and zinc (0.120 mg/L) (U.S. Environmental Protection Agency 2002). The CCC limits indicate potential for adverse effects resulting from long-term (30-day) exposure. Although storm conditions lasting only hours to days accounted for most exceedances of water-quality criteria, impounding the storm water could prolong exposure (Fishel 1988).

Metal-rich suspended solids and streambed sediments represent a potential source of dissolved metals. Solid forms of the metals could be ingested by aquatic organisms with subsequent uptake of dissolved species within the gut. Dissolved metals also could be derived by recrystallization of metastable solid phases to more stable phases (Bigham and Nordstrom 2000), by dissolution or desorption (Francis et al. 1989; Webster et al. 1998), and/or by reductive dissolution of Fe^{III} and Mn^{III-IV} oxides (Francis and Dodge 1990). These processes could be promoted by decreases in pH and/or redox potential in the streambed or water column.

Twenty-four of the 33 fish species identified in Swatara Creek at Ravine during the study had been previously reported for Pennsylvania streams with pH 4.6 to 6.4 (Table 6). A subset of these fish was found in Good Spring Creek at Tremont, Lorberry Creek at Lorberry Junction, and Swatara Creek at Newtown (Table 6). According to Earle and Callaghan (1998), only 18 species of fish native to Pennsylvania have been found in Pennsylvania streams having pH <6; the majority of these species now can be found in Swatara Creek.

Concentrations of dissolved sulfate, iron, and manganese were greater for Lorberry Creek and Good Spring Creek than Swatara Creek at Ravine or Swatara Creek at Newtown (Figure 21). Although Good Spring Creek and Lorberry Creek had fewer fish than Swatara Creek at Ravine, these sites had more fish than Swatara Creek at Newtown (Figure 20). Such differences in fish numbers and species diversity probably reflect smaller streamflows and limited habitat at the upstream sites. Sections of the surveyed reach at Newtown flowed intermittently during the study. Generally, greater species diversity and larger populations would be expected for larger aquatic habitats (e.g. McNicol 2002). Although fish surveys were not conducted prior to 2002 for Lorberry Creek, potential downward trends in acidity and dissolved metals concentrations for Lorberry Creek during the study (Figure 21) could explain the appearance of blacknose dace, creek chub, and brook trout in this tributary. These species, which are moderately tolerant of low pH and pollution (Table 6), were among the first species found in Swatara Creek at Ravine during 1996, indicating early stages of its ecological recovery. Similarly tolerant fish species have been identified as early colonists in other systems recovering from acidification (e.g. Cravotta 2005; Mills et al. 2000; Short et al. 1990).

Cravotta and Bilger (2001) presented results for macroinvertebrate surveys on Swatara Creek at Ravine conducted during 1996-2000. Although such data were not collected during 2001 to 2006, a macroinvertebrate survey of Swatara Creek at Ravine in September 2007 indicated results comparable to previous assessments (Gary Walters, Pennsylvania Department of Environmental Protection, 2008, written commun.). Although 11 benthic macroinvertebrate taxa (family level), including 3 genera of Ephemeroptera (mayflies) were found in Swatara Creek at Ravine in 2000, a few relatively pollution- tolerant taxa dominated. More than half of the individual specimens identified in 2000 were Hydropsyche and Chironomidae, which are tolerant of metals and acidic conditions (e.g. Chadwick and Canton 1986; Courtney and Clements 2002; Short et al. 1990; Tomkiewicz and Dunson 1977). The lack of taxa richness and trophic imbalance in Swatara Creek is consistent with the identified toxic effect levels for metals in the streambed sediments (Cravotta and Bilger 2001) and implies that metals in the aquatic environment that are stressful to macroinvertebrates may not be severely limiting to fish. Because native fish populations had returned, but the macroinvertebrate community continued to indicate water-quality impairment in 2007, Swatara Creek was characterized as “partially meeting designated uses” and was not removed from the 2008 Pennsylvania Integrated Water Quality Monitoring and Assessment Report as impaired (Pennsylvania Department of Environmental Protection 2007).

Summary and Conclusions

A variety of treatment systems (including some land reclamation) was installed for the neutralization of acidity and the removal of dissolved metals from AMD sources and downstream sites in Swatara Creek and its tributaries; eight systems evaluated in this paper were installed in 1995-2001. Periodic measurements of flow rate and chemical concentrations upstream and downstream of each system indicated that all eight treatment systems evaluated in this paper were effective at decreasing the acidity load. However, each system had unique characteristics, and the treatment performance varied considering the acid load removed relative to the size of the treatment system and the cost of treatment. Generally, the treatment costs were consistent with results of other treatment systems presented by Ziemkiewicz et al. (2003). In summary, (1) the limestone- sand dosing was relatively simple and inexpensive to implement and had positive water-quality effects; (2) the open limestone channel generally had negligible effects on water quality and was relatively expensive; (3) the oxic limestone drain removed significantly more acidity than the limestone sand treatment but was relatively inefficient considering the amount and cost of the limestone used; (4) the anoxic limestone drain was effective at removing acidity and at relatively low cost; (5) the two sets of limestone diversion wells were relatively expensive but effective for treating streamwater during high-flow conditions; and (6) the aerobic wetlands and limestone-compost based wetlands generally were effective at attenuating dissolved and suspended metals during base-flow conditions but were less effective during stormflow conditions. Generally, stormflow can be acidic, and, as streamflow volume increases, a smaller fraction of total flow tends to be treated and (or) residence time in the treatment system will be reduced. Furthermore, during stormflow conditions, metal-rich sediments commonly can be scoured and resuspended from the streambed.

Generally, to maintain neutral pH during storms, additional limestone diversion wells could be constructed to begin or increase alkalinity production as the stream stage rises and/or additional or larger limestone drains could be constructed to produce greater amounts of alkalinity and enhance the buffering capacity of base flow. Nevertheless, neutralization and pH buffering alone will not remedy the problem of metals transport. Alkalinity-producing systems such as limestone diversion wells or limestone drains combined with wetlands could be needed to attenuate metals transport. Because of potential adverse effects on water temperature, designs for constructed wetlands and other treatments should consider factors such as shading, aspect, water depth, and retention time, all of which can affect temperature.

Monitoring of the untreated influent, treated effluent, and associated changes in streamwater quality over a range of hydrologic conditions is needed to indicate treatment-system performance and environmental benefits. To indicate long-term performance of treatment systems, monitoring and documentation of treatment-system maintenance are needed for the duration of the anticipated service life. Given such long-term data, performance metrics, such as the average acid load removed as a function of treatment system size or cost, can be improved and considered by resource managers and other stakeholders involved in mine-drainage remediation.

Streams affected by “acidic” mine drainage (AMD) in the northeastern USA commonly have diminished fish populations because of aquatic habitat degradation associated with low pH and/or elevated concentrations of iron, aluminum, and other metals from the AMD. Nevertheless, as impacts from AMD become less severe through natural attenuation and/or watershed-restoration activities, fish populations may rebound. For example, upper Swatara Creek, which drains the Southern Anthracite Coalfield in eastern Pennsylvania, had been contaminated by AMD for most of the 20th century. Nevertheless, because of progressive improvement in water quality and the recovery of native fish populations described in this paper, upper Swatara Creek recently (2008) was characterized by the Pennsylvania Department of Environmental Protection (2008) as “partially meeting designated uses” and by the U.S. Environmental Protection Agency (2007) as a “nonpoint-source success story.”

More than four decades of intermittent monitoring of base flow of Swatara Creek immediately downstream from the mined area indicated median sulfate concentration decreased from about 150 mg/L in 1959 to 50 mg/L in 2007; pH increased from acidic to near-neutral values (medians: pH~4 before 1975; pH~6.5 after 1995). These long-term trends probably resulted from a decrease in pyrite oxidation and the onset of carbonate buffering, partly because of flooding the mines during the early period and the dissolution of limestone in treatment systems during the later period. As a consequence of the improved water quality, fish populations in Swatara Creek rebounded from nonexistent during 1959-90 to as many as 25 species during 1996-2006, including several taxa that are intolerant of low pH and pollution.

The AMD treatments with limestone that were implemented during 1995-2001 in the upper part of the Swatara Creek watershed added alkalinity, which was needed to maintain near-neutral pH, and calcium, which can be important to aquatic organisms for regulating toxic effects of dissolved metals. The treatments not only reduced the influence of AMD, but also mitigated extreme fluctuations in pH of Swatara Creek immediately downstream from the mined area that were associated with episodic acidification during storm runoff events. During 1996-1998, pH values approaching 5.0 were frequently recorded during stormflow events; however, during 1999-2007, after treatments were implemented in the upper watershed, such low-pH excursions were rarely recorded.

Sulfate concentration, SC, and pH of Swatara Creek at Ravine were inversely correlated with streamflow because of dilution of poorly buffered stream water with weakly acidic storm runoff. In contrast, total and dissolved concentrations of metals were poorly correlated with streamflow because concentrations of suspended solids and metals typically peak prior to peak stream stage (hysteresis). As a result of scour and transport of the metals in streambed sediments, concentrations of suspended solids and total metals in the water column are correlated, and those for stormflow typically exceed base flow.

Despite near-neutral, aerobic, cool-water conditions in Swatara Creek that support a diverse fish population, untreated AMD and metal-rich streambed sediments in this and other mining-affected watersheds represent a substantial, long-term source of metals that are likely to impair water quality and complicate aquatic ecological recovery for the future. Although the transport of dissolved iron, aluminum, and most trace metals typically is attenuated at near-neutral pH, substantial transport of suspended and dissolved metals persists in Swatara Creek, especially during stormflow conditions. Iron, aluminum, and, to a lesser extent, manganese, nickel, and zinc, are transported as suspended particles resulting from scour and transport of metal-enriched streambed deposits. Total iron, manganese, aluminum, and associated trace metals commonly increase in concentration at the onset of stormflow conditions; peak metal concentrations typically are achieved prior to peak discharge. The metal content of the suspended solids is relatively constant over the range of streamflow conditions, implying a relatively uniform source of material such as streambed deposits.

On the basis of combined methods using fixed-interval base-flow and automated stormflow sampling, total concentrations and loads of suspended sediment and metals were shown to be greatest and pH lowest during stormflow conditions in Swatara Creek. In general, temporal variations in water quality of low- order streams such as the northern part of Swatara Creek are difficult to characterize by routine monitoring at fixed-time intervals. This routine works well to characterize base-flow conditions and to establish potential long-term trends but is not appropriate to characterize rapidly changing conditions in response to streamflow. Automated samplers and continuous water-quality and streamflow monitoring methods, as used in this study, generally will indicate extremes, which can be important with respect to biological or regulatory thresholds, and can indicate significant relations between streamflow, water chemistry, and transport of sediment and associated chemicals. Water-quality regulations established to achieve in-stream water-quality standards or to maintain designated uses of the water body (water supply, fishing, etc.), such as Total Maximum Daily Loads (TMDLs), require baseline characterization of pollutant loads in order to determine required reductions in loading from various contaminant sources (Caruso 2005; Pennsylvania Department of Environmental Protection 1999, 2002). Data that do not adequately represent stormflow conditions will underestimate the transport of sediment and associated metals and will not be useful to establish the data distribution.

Generally, to maintain stream pH in water bodies affected by AMD and subject to acidification during storms, limestone diversion wells or other dosing systems could be constructed to begin or increase alkalinity production as the stream stage rises, and underground limestone drains and/or limestone-filled basins could be constructed at AMD sites to enhance the buffering capacity of base flow. Nevertheless, as the limestone in treatment systems is consumed, supplemental buffering capacity would be needed to maintain near-neutral pH. Furthermore, neutralization and pH buffering alone will not remedy the problem of metals transport. Solid forms of the metals, as particulate and particle coatings, can be ingested and accumulated by aquatic organisms and can be remobilized by reductive dissolution of Fe^{III} and Mn^{III-IV} oxides in buried sediment. Additional measures such as wetlands and holding basins for stormwater could be warranted to prevent metals transport to the stream. However, impounding water in wetlands and shallow ponds could increase warming of the water during summer, potentially leading to temperatures that are not favorable to fish. If the restoration of fish and other aquatic organisms is a goal of reclamation, strategies for AMD treatment should be considered that minimize the potential for excessive warming of the water while removing toxic metals.

Although the study demonstrated that positive streamwater-quality trends and the recovery of fish populations in Swatara Creek coincided with implementation of AMD treatments in the upper watershed, the study was not designed to identify specific hydrochemical thresholds that were critical to aquatic ecological recovery. Streamwater-quality data were collected continuously during the study; however, fish population data were collected only annually. Furthermore, the potential for synergistic effects of various toxic chemical constituents combined with changes in water temperature complicate the interpretations of ecological stressors. Supplemental data on the populations of fish and other aquatic organisms before and after extreme hydrologic events (droughts, storms) coupled with water-quality data for such events would be needed to establish relations between transient water-quality conditions and specific factors that could be limiting to the aquatic organisms.

Final Remarks

Two technical papers were written by Charles A. Cravotta III, USGS to summarize the work completed by this project. "Abandoned Mine Drainage in the Swatara Creek Basin, Southern Anthracite Coalfield, Pennsylvania, USA: 1. Streamwater-Quality Trends Coinciding with the Return of Fish" and "Abandoned Mine Drainage in the Swatara Creek Basin, Southern Anthracite Coalfield, Pennsylvania, USA: 2. Performance of Treatment Systems". This report largely consists of excerpts from these papers. Acknowledgements can be found in these technical papers. Many other articles and posters were developed from the monitoring for this project in the Swatara Creek Watershed. They can be found in Appendix B.

Public Involvement

The effort to improve the water quality of Swatara Creek included dedication of resources and funding from the federal and state agencies, grass roots commitment of citizens and local government, and cooperation of industry. Throughout the 1990's federal funding was provided through the EPA 104(b)3 and the EPA 319 program and state funding was provided through Growing Greener and DEP Reclamation-in-lieu of Civil Penalties. The Schuylkill Conservation District took the lead as the grant applicant and coordinator of the restoration effort and the projects are documented in this paper. The DEP Pottsville District Mining also played a significant role in directing funding and resources to the effort. Most of the projects were focused on treatment or prevention of mine drainage. The local mining industry assisted on several projects by providing machinery and manpower for completion of project as well as maintenance.

The efforts to clean up the watershed were originally driven by government agencies, but it became evident early on that the citizens were anxious to be willing participants. The Northern Swatara Creek Watershed Association formed in 1996 to involve the public in the newly started effort to acquire funding to address the numerous sources of pollution. The Schuylkill County Conservation District spearheaded the effort to obtain funding. In the beginning many of the citizens saw the impairments as obstacles too difficult to overcome. The citizens' goal from the onset was to restore Swatara Creek to fishable waters. As projects were completed throughout the 1990's the citizens played a more active role in directing and maintaining the projects. As the improvements in Swatara Creek were realized the citizens and the watershed association began to take advantage of the recreational value of the improved quality of the creek. They formed the Northern Swatara Cooperative Trout Nursery to raise trout to stock in the much improved Swatara Creek. Trout have been stocked annually in Swatara Creek and its tributaries by the watershed association since 1999 and fishing has been restored as a recreational pastime for the first time many years.

Lessons Learned

For a project this long and this complex there are many issues that arise that must be dealt with. They all provide a learning experience. The following are some of these lessons learned.

Scope: Inflow-outflow and upstream-downstream monitoring was originally planned for a total of 20 sites for 3 years. However, because of apparent improvement in downstream water quality that could be short lived and because additional treatment systems were installed or planned that would have downstream effects at Ravine, the project term was extended twice (3- and 5-years). To evaluate the additional treatment systems, the number of monitoring sites nearly tripled. Because of the added monitoring workload, the frequency of sampling at most sites had to be reduced from monthly to six-week intervals or bimonthly. Furthermore, monitoring was discontinued at some sites where treatment effects were not apparent (e.g. Coal Run limestone sand) or where treatment system maintenance was inadequate (e.g. Orchard oxic limestone drain, Martin Run diversion wells).

Equipment: To evaluate water-quality and chemical transport during high-flow events, storm samplers and continuous water-quality monitoring were used along with continuous stage/flow recorders at selected sites. This "automated" approach worked extremely well, but was very labor intensive and expensive. The dedicated monitoring equipment required frequent maintenance (4-to-6-week intervals) and occasional replacement because of lightning strikes or other technical problems. By the last half of the project, the use of automated storm sampling and water-quality monitoring was pared down to a couple sites to ensure data quality with limited budget.

Lab analyses: Over the 11-year project period, laboratory analyses were provided by PaDEP, DOE, and Actlabs. Although the same general procedures were used by each lab, differences in services provided, detection limits, and reporting procedures complicated the data base management and data analysis. Costs for lab analysis increased over time, but the project budget did not include sufficient inflation factor. The increase in annual lab cost coupled with increased number of monitoring sites required reductions in scope to stay within budget. These changes complicated data analysis.

Labor: Over the 11-year project period, staffing for field work changed too frequently. In some cases, the experienced staff member was not available to train their replacement. Documentation of equipment maintenance and commitment to collect good quality data varied among participants.

References

American Public Health Association (1998a) Alkalinity (2320)/Titration method. In: Clesceri LS, Greenberg AE, Eaton AD (Eds), Standard Methods for the Examination of Water and Wastewater, 20th edit, American Public Health Assoc, Washington, DC, p 2.26-2.30

American Public Health Association (1998b) Acidity (2310)/Titration method. In: Clesceri LS, Greenberg AE, Eaton AD (Eds), Standard Methods for the Examination of Water and Wastewater, 20th edit, American Public Health Assoc, Washington, DC, p. 2.24-2.26

Anthracite Research and Development Co., Inc. (1972) Swatara Creek mine drainage pollution abatement project, part three, Operation Scarlift. Commonwealth of Pennsylvania SL-126-3, 113 pp

Arnold DE (1991) Diversion wells--a low-cost approach to treatment of acid mine drainage. In Proceedings of the 12th Annual West Virginia Surface Mine Drainage Task Force Symposium. Charleston, W.Va., West Virginia Mining and Reclamation Association, p. 39-50

Baker JP, Schofield CL (1982) Aluminum toxicity to fish in acidic waters. Water Air Soil Pollution 18:289-309

Barbour MT, Gerritsen J, Snyder BD, Stribling JB (1999) Rapid bioassessment protocols for use in streams and wadeable rivers--periphyton, benthic macroinvertebrates, and fish. 2nd edit, US EPA 841-B-99-002, 11 ch, 4 appendices (www.epa.gov/OWOW/monitoring/ techmon.html)

Berg TM, Barnes JH, Severn WD, Skema VK, Wilshusen JP, Yannaci DS (1989) Physiographic provinces of Pennsylvania. Pennsylvania Geol Surv Map 13, scale 1:2,000,000

Berger Associates, Inc. (1972) Swatara Creek mine drainage pollution abatement project, part two, Operation Scarlift. Commonwealth of Pennsylvania, SL-126-2, 168 pp

Bigham JM, Nordstrom DK (2000) Iron and aluminum hydroxysulfate minerals from acid sulfate waters. In: Jambor JL, Alpers CN, Nordstrom DK (Eds), Sulfate minerals, crystallography, geochemistry and environmental significance, Mineralogical Soc of America Reviews in Mineralogy and Geochemistry 40: 351-403

Blowes DW, Ptacek CJ, Jambor JL, Weisener CG (2003) The geochemistry of acid mine drainage. In Lollar, B.S. (Ed), Environmental Geochemistry. Elsevier, Treatise on Geochemistry 9:149-204

Bowes MJ, House WA, Hodgkinson RA, Leach DV (2005) Phosphorus-discharge hysteresis during storm events along a river catchment: the River Swale, UK. Water Research 39:751-762

Bradford AD, Sickles WN (1950) Swatara Creek, tributary to Susquehanna River Schuylkill, Lebanon, Dauphin Counties. Pennsylvania Fish and Boat Commission Stream Survey Report, October 16-19, 1950

Broshears RE, Runkel RL, Kimball BA, McKnight DM, Bencala KE (1996) Reactive solute transport in an acidic stream--experimental pH increase and simulation of controls on pH, aluminum, and iron. Environ Sci Technol 30:3016-3024

Burrows WD (1977) Aquatic aluminum--chemistry, toxicology, and environmental prevalence. CRC Critical Reviews in Environmental Controls 7:167-216

Butler RL, Cooper EL, Crawford JK, Hales DC, Kimmel WG, Wagner CC (1973) Fish and food organisms in acid mine waters of Pennsylvania. US EPA-R3-73-032, 158 pp

Cannon WE, Kimmel WG (1992) A comparison of fish and macroinvertebrate communities between an unpolluted stream and the recovery zone of a stream receiving acid mine drainage. J Pennsylvania Academy of Science 66:58-62

Caruso BS (2005) Simulation of metals total maximum daily loads and remediation in a mining-impacted stream. Jour Environ Eng 131:777–789

Caruso BS, Cox TJ, Runkel RL, Velleux ML, Bencala KE, Nordstrom DK, Julien PY, Butler BA, Alpers CN, Marion A, Smith KS (2008) Metals fate and transport modelling in streams and watersheds: state of the science and USEPA workshop review. Hydrological Processes 22:4011-4021

Chadwick JW, Canton SP (1986) Recovery of benthic macroinvertebrate communities in Silver Bow Creek, Montana, following improved metal mine wastewater treatment. Water Air Soil Pollut 28:427-438

Cleveland L, Buckler DR, Brumbaugh WG (1991) Residue dynamics and effects of aluminum on growth and mortality in brook trout. Environ Toxicol Chem 10:243-248

Cohn TA (1988) Adjusted maximum likelihood estimation of the moments of lognormal populations from type 1 censored samples. US Geol Surv Open-File Report 88-350, 34 pp

Cohn TA, DeLong LL, Gilroy EJ, Hirsch RM, Wells RM (1989) Estimating constituent loads. Water Resour Research 25:937-942

Commonwealth of Pennsylvania (2002) Chapter 93. Water quality standards. Pennsylvania Code, Title 25. Environmental Protection. Harrisburg, Pa. Commonwealth of Pennsylvania p. 93.1-93.226

Corbett ES, Lynch JA (1982) Rapid fluctuations in streamflow pH and associated water-quality parameters during a stormflow event. In: International Symposium on Hydrometeorology. American Water Resources Association p. 461-464

Coston JA, Fuller CC, Davis JA (1995) Pb^{2+} and Zn^{2+} adsorption by a natural aluminum- and iron-bearing surface coating on an aquifer sand. Geochim Cosmochim Acta 59:3535-3547.

Courtney LA, Clements WH (2002) Assessing the influence of water and substratum quality on benthic macroinvertebrate communities in a metal-polluted stream--an experimental approach. Freshwater Biology 47:1766–1778

Cravotta CA III (1994) Secondary iron-sulfate minerals as sources of sulfate and acidity--The geochemical evolution of acidic ground water at a reclaimed surface coal mine in Pennsylvania. In Environmental geochemistry of sulfide oxidation, eds. Alpers, C.N. and D.W. Blowes. American Chemical Society Symposium Series 550, p. 345-364

Cravotta CA III (2000) Relations among sulfate, metals, sediment, and streamflow data for a stream draining a coal-mined watershed in east-central Pennsylvania. In: ICARD 2000 Proceedings Fifth International Conference on Acid Rock Drainage. Littleton, Co., Society for Mining, Metallurgy, and Exploration, Inc. 1:401-410

Cravotta CA III (2003) Size and performance of anoxic limestone drains to neutralize acidic mine drainage. J Environ Qual 32:1277-1289

Cravotta CA III (2005) Effects of abandoned coal-mine drainage on streamflow and water quality in the Mahanoy Creek Basin, Schuylkill, Columbia, and Northumberland Counties, Pennsylvania, 2001. US Geol Surv Scientif Investigat Rep 2004-5291, 60 pp

Cravotta CA III (2007) Passive aerobic treatment of net-alkaline, iron-laden drainage from a flooded underground anthracite mine, Pennsylvania, USA. Mine Water Environ 26:128-149

Cravotta CA III (2008a) Dissolved metals and associated constituents in abandoned coal-mine discharges, Pennsylvania, USA: 2. Geochemical controls on constituent concentrations. Appl Geochem 23:203-226

Cravotta CA III (2008b) Laboratory and field evaluation of a flushable oxic limestone drain for treatment of net-acidic, metal-laden drainage from a flooded anthracite mine, Pennsylvania, USA. Appl Geochem 23:3404-3422

Cravotta CA III (2009) Abandoned mine drainage in the Swatara Creek Basin, Southern Anthracite Coalfield, Pennsylvania, USA: 1. Streamwater-quality trends coinciding with the return of fish. Mine Water Environ (in press)

Cravotta CA III (2009) Abandoned mine drainage in the Swatara Creek Basin, Southern Anthracite Coalfield, Pennsylvania, USA: 2. Performance of passive-treatment systems. Mine Water Environ (in press)

Cravotta CA III, Bilger MD (2001) Water-quality trends for a stream draining the Southern Anthracite Field, Pennsylvania. Geochem Explor Environ Anal 1:33-50

Cravotta CA III, Brady KBC, Rose AW, Douds JB (1999) Frequency distribution of the pH of coal-mine drainage in Pennsylvania. In U.S. Geological Survey Toxic Substances Hydrology Program--Proceedings of the Technical Meeting, Charleston, South Carolina, March 8-12, 1999, Morganwalp DW, Buxton H (Eds). US Geol Surv Water-Resour Inv Rep. 99-4018A, p. 313-324

Cravotta CA III, Kirby, CS (2004) Effects of abandoned coal-mine drainage on streamflow and water quality in the Shamokin Creek Basin, Northumberland and Columbia Counties, Pennsylvania, 1999-2001. US Geol Surv Water-Resour Inv Rep 03-4311, 58 pp

Cravotta CA III, Nantz, JM (2008) Quantity and quality of stream water draining mined areas of the upper Schuylkill River basin, Schuylkill County, Pennsylvania, 2005-2007: In, National Meeting of the American Society of Mining and Reclamation, Richmond VA, New Opportunities to Apply Our Science June 14-19, 2008. American Society of Mining and Reclamation, p. 223-252

Cravotta CA III, Trahan MK (1999) Limestone drains to increase pH and remove dissolved metals from acidic mine drainage. Appl Geochem 14:581-606

Cravotta CA III, Ward SJ, Koury DJ, Koch RD (2004) Optimization of limestone drains for long-term treatment of acidic mine drainage, Swatara Creek Basin, Schuylkill County, PA. Proc, 2004 National Meeting of the American Soc of Mining and Reclamation and the 25th WV Surface Mine Drainage Task Force, p 366-411

Cravotta CA III, Weitzel JB (2001) Detecting change in water quality from implementation of limestone treatment systems in a coal-mined watershed, Pennsylvania. In: Proceedings 8th National Nonpoint Source Monitoring Program Workshop. USEPA Seminar Series EPA/905-R-01-008

Crock JG, Arbogast BF, Lamothe PJ (1999) Laboratory methods for the analysis of environmental samples. In: Plumlee GS, Logsdon MJ (Eds), The Environmental Geochemistry of Mineral Deposits--Part A. Processes, Techniques, and Health Issues, Soc of Economic Geologists, Reviews in Economic Geology, 6A, p 265-287

DeWalle DR (1990) Atmospheric deposition effects on aquatic chemistry in Pennsylvania--watershed processes and current state of knowledge. In: Lynch, JA, Corbett, ES, Grimm, JW (Eds) Atmospheric Deposition in Pennsylvania--A critical assessment. The Pennsylvania State University Environmental Resources Research Institute p. 63-81

Dsa JV, Johnson KS, Lopez D, Kanuckel C, Tumulinson J (2008) Residual toxicity of acid mine drainage- contaminated sediment to stream macroinvertebrates--Relative contribution of acidity vs. metals. Water Air Soil Pollut 194:185-197

Earle J, Callaghan T (1998) Effects of mine drainage on aquatic life, water uses, and man-made structures. In: Brady KBC, Smith MW, Schueck JH (Eds), The prediction and prevention of acid drainage from surface coal mines in Pennsylvania, Pennsylvania Dept of Environmental Protection, Harrisburg, PA, USA, 5600-BK- DEP2256, p 4.1-4.10

Eggleston JR, Kehn TM, Wood GH Jr (1999) Anthracite. In: Schultz CH (Ed), The Geology of Pennsylvania. PA Geological Survey. 4th series, Special Publ 1, p 458-469

Elder JF (1988) Metal biogeochemistry in surface-water systems--a review of principles and concepts. US Geol Surv Circular 1013, 43 p

Ficklin WH, Mosier EL (1999) Field methods for sampling and analysis of environmental samples for unstable and selected stable constituents. In: Plumlee GS, Logsdon MJ (Eds), The Environmental Geochemistry of Mineral Deposits--Part A. Processes, Techniques, and Health Issues, Soc of Economic Geologists, Reviews in Economic Geology, 6A:249-264

Findlay D (2003) Response of phytoplankton communities to acidification and recovery in Killarney Park and the Experimental Lakes Area, Ontario. Ambio 32:190-195

Fishel DK (1988) Preimpoundment hydrologic conditions in the Swatara Creek (1981-84) and estimated postimpoundment water quality in and downstream from the planned Swatara State Park reservoir, Lebanon and Schuylkill Counties, Pa. US Geol Surv Water-Resour Inv Rep 88-4087, 108 p

Fishman MJ, Friedman LC (Eds) (1989) Methods for determination of inorganic substances in water and fluvial sediments. US Geol Surv Tech Water-Resour Inv 05-A1, 545 p

Francis AJ, Dodge DJ, Rose AW, Ramirez AJ (1989) Aerobic and anaerobic microbial dissolution of toxic metals from coal wastes--mechanism of action. Env Sci Tech 23:435-441

Francis AJ, Dodge DJ (1990) Anaerobic microbial remobilization of toxic metals coprecipitated with iron oxide. Environ Sci & Technol 24:373-378

Galloway JN (2001) Acidification of the world. Water Air Soil Pollut 130:17–24

Gannett Fleming Corddry and Carpenter, Inc. (1972) Swatara Creek mine drainage abatement project, part 1, Operation Scarlift: Commonwealth of Pennsylvania SL-126-1, 57 pp

Gray NF (1997) Environmental impact and remediation of acid mine drainage--a management problem. Environ Geol 30:62-71

Growitz DJ, Reed LA, Beard MM (1985) Reconnaissance of mine drainage in the coal fields of eastern Pennsylvanian. US Geol Surv Water-Resour Inv Rep 83-4274, 54 pp

Havas M, Rosseland BO (1995) Response of zooplankton, benthos, and fish to acidification--an overview. Water Air Soil Pollut 85:51-62

Hedin RS, Nairn RW, Kleinmann RLP (1994a) Passive treatment of coal mine drainage. USBM IC 9389, 35 pp

Hedin RS, Watzlaf GR, Nairn RW (1994b) Passive treatment of acid mine drainage with limestone. J Environ Qual 23:1338-1345

Helsel DR, Hirsch RM (2002) Statistical methods in water resources. US Geol Surv Techniques of Water- Resources Investigations 04-A3, 523 p

Henry TB, Irwin ER, Grizzle JM, Wildhaber ML, Brumbaugh WG (1999) Acute toxicity of an acid mine drainage mixing zone to juvenile bluegill and largemouth bass. Trans American Fisheries Soc 128:919-928

Herlihy AT, Kaufmann PR, Mitch ME, Brown DD (1990) Regional estimates of acid mine drainage impact on streams in the mid-Atlantic and southeastern United States. Water Air Soil Pollut 50:91-107

Herricks, EE (1977) Recovery of streams from chronic pollutional stress--Acid mine drainage. In: Cairns J Jr, Dickson KL, Herricks EE (Eds), Recovery and restoration of damaged ecosystems. University Press of Virginia, Charlottesville VA, p. 43-71

Hoffman GL, Fishman MJ, Garbarino JR (1996) Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory--In-bottle acid digestion of whole-water samples. US Geol Surv Open- File Rep. 96-225, 28 p

Holt C, Yan ND (2003) Recovery of crustacean zooplankton communities from acidification in Killarney Park, Ontario, 1971–2000: pH 6 as a recovery goal. Ambio 32:203-207

Jackson LR (1987) Swatara Creek, subbasin 07, subsubasin D (river miles 58.25 to 4.6). Pennsylvania Fish and Boat Commission Stream Survey Report, July 15, 1987

Johnson DB, Hallberg KB (2005) Acid mine drainage remediation options: a review. Sci Total Environ 338:3-14

Kairies CL, Capo RC, Watzlaf GR (2005) Chemical and physical properties of iron hydroxide precipitates associated with passively treated coal mine drainage in the Bituminous Region of Pennsylvania and Maryland. Appl Geochem 20:1445-1460

Kimball BA, Callender E, Axtmann EV (1995) Effects of colloids on metal transport in a river receiving acid mine drainage, upper Arkansas River, Colorado, USA. Appl Geochem 10:285-306

Kirby CS, Cravotta CA III (2005) Net alkalinity and net acidity 2: Practical considerations. Appl Geochem 20:1941-1964

Koury DJ, Hellier WW (1999) Constructed wetland for mine drainage treatment Lorberry Junction wetland project. National Assoc Abandoned Mine Lands Programs Conference Pennsylvania, August 22- 25, 1999, 9 pp

Langland MJ, Raffensperger JP, Moyer DL, Landwehr JM, Schwarz GE (2006) Changes in streamflow and water quality in selected nontidal basins in the Chesapeake Bay watershed, 1985-2004. US Geol Surv Scientif Investigat Rep 2006-5178, 75 pp

McCarren EF, Wark JW, George JR (1964) Water quality of the Swatara Creek Basin, Pa. US Geol Surv Open-File Rep., 88 p

MacDonald DD, Ingersoll CG, Berger TA (2000) Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Arch Environ Contam Toxicol 39:20-31

McNicol DK (2002) Relation of lake acidification and recovery to fish, common loon, and common merganser occurrence in Algoma lakes. Water Air Soil Pollut Focus 2:151–168

Mills KH, Chalanchuk SM, Allan DJ (2000) Recovery of fish populations in Lake 223 from experimental acidification. Can J Fish Aquat Sci 57:192–204

Monteith DT, Hildrew AG, Flower RJ, Raven PJ, Beaumont WRB, Collen P, Kreiser AM, Shilland EM, Winterbottom JH (2005) Biological responses to the chemical recovery of acidified fresh waters in the UK. Environmental Pollution 137:83-101

Nelson SM, Roline RR (1996) Recovery of a stream macroinvertebrate community from mine drainage disturbance. Hydrobiologia 339:73-84

Nordstrom DK (2000) Advances in the hydrochemistry and microbiology of acid mine waters. International Geology Review 42:499-515

Olyphant GA, Bayless ER, Harper D (1991) Seasonal and weather-related controls on solute concentrations and acid drainage from a pyritic coal-refuse deposit in southwestern Indiana, U.S.A. J Contam Hydrol 7:219-236

Pennsylvania Department of Environmental Protection (1999) Upper Swatara Creek watershed TMDL for acid mine drainage affected segments. PA Dept of Environmental Protection, Bureau of Watershed Mgmt, Harrisburg, PA, 26 pp. (http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/tmdl/ Swat_TMDL.pdf)

Pennsylvania Department of Environmental Protection (2000) Erosion and sediment pollution control program manual. PA Dept of Environmental Protection, Bureau of Watershed Mgmt, Harrisburg, PA, Doc # 363-2134-008, tables 9 and 10A

Pennsylvania Department of Environmental Protection (2002) Watershed management and TMDL's. PA Dept of Environmental Protection, Bureau of Watershed Mgmt, Harrisburg, PA, Fact Sheet 3800-FS- DEP2248 Rev. 1/2002, 4 pp. (http://www.dep.state.pa.us/dep/deputate/watermgt/wqp/wqstandards/Facts/ fs2248.pdf)

Pennsylvania Department of Environmental Protection (2004) Watershed restoration action strategy (WRAS), State water plan subbasin 07D Swatara Creek watershed, Dauphin, Lebanon, Berks, and Schuylkill Counties. PA Dept of Environmental Protection, Bureau of Watershed Mgmt, Harrisburg, PA, 47 pp. (http://www.depweb.state.pa.us/watershedmgmt/lib/watershedmgmt/nonpoint_source/wras/wras-07d.pdf)

Pennsylvania Department of Environmental Protection (2007) 2006 Pennsylvania integrated water quality monitoring and assessment report--Clean Water Act Section 305(b) report and 303(d) list. PA Dept of Environmental Protection, Bureau of Watershed Mgmt, Harrisburg, PA, 55 pp. (http:// www.depweb.state.pa.us/watersupply/lib/watersupply/MasterNarrative.pdf)

Pennsylvania Department of Environmental Protection (2008) Pennsylvania nonpoint source management program FFY2007 Annual Report--Measuring project and program effectiveness, October 1, 2006 through September 30, 2007. PA Dept of Environmental Protection, Bureau of Watershed Mgmt, Harrisburg, PA, 104 pp. (http://www.depweb.state.pa.us/watershedmgmt/lib/watershedmgmt/nonpoint_source/initiatives/2007_nps_annual_report_entire_document.pdf)

Pennsylvania Organization for Watersheds and Rivers (2002) Abandoned mine reclamation in Pennsylvania--The Abandoned Mine Reclamation Fund, partnerships, and future challenges. Pennsylvania Organization for Watersheds and Rivers, Watershed Fact Pack, 28 pp. (http://www.pawatersheds.org/ factpack/AbandonedMine.pdf)

Rantz SE and others (1982a) Measurement and computation of streamflow--1. Measurement of stage and discharge. US Geol Surv Water-Supply Paper 2175, v 1, 284 pp

Rantz SE and others (1982b) Measurement and computation of streamflow--2. Computation of discharge. US Geol Surv Water-Supply Paper 2175, v 2, 631 pp

Raymond PA, Oh N-H (2009) Long-term changes of chemical weathering in rivers heavily impacted from acid mine drainage: Insights on the impact of coal mining on regional and global carbon and sulfur budgets. Earth Planet Sci Let 284:50–56

Rose AW, Cravotta CA III (1998) Geochemistry of coal-mine drainage. In: Brady KBC, Smith MW, Schueck JH (Eds), The prediction and prevention of acid drainage from surface coal mines in Pennsylvania. PA Dept of Environmental Protection, Harrisburg, PA, USA, 5600-BK-DEP2256, p 1.1- 1.22

Rutledge AT (1998) Computer programs for describing the recession of ground-water discharge and for estimating mean ground-water recharge and discharge from streamflow data - update. US Geol Surv Water-Resour Inv Rep 98-4148, 43 pp

Schemel LE, Kimball BA, Bencala KE (2000) Colloid formation and metal transport through two mixing zones affected by acid mine drainage near Silverton, Colorado. Appl Geochem 15:1003-1018

Schuylkill Conservation District, Schuylkill Headwaters Association, Inc. and RETTEW Associates, Inc. (2006) Upper Swatara Creek TMDL watershed implementation plan, Schuylkill County. PA Dept of Environmental Protection, Bureau of Watershed Mgmt, Harrisburg, PA, 50 pp. (http:// www.depweb.state.pa.us/watershedmgmt/lib/watershedmgmt/nonpoint_source/implementation/final_upper_swatara_creek_ implementation_plan.pdf)

Shoemaker ME (1932) Swatara Creek, Schuylkill County. Pennsylvania Fish and Boat Commission Stream Survey Report, January 15, 1932

Short TM, Black JA, Birge WJ (1990) Effects of Acid-mine drainage on the chemical and biological character of an alkaline headwater stream. Arch Environ Contam Toxicol 19:241-248

Skelly & Loy, Inc. (1987) A watershed pollution study of the Swatara Creek. PA Dept of Environmental Resources, Bureau of Abandoned Mine Reclamation, Final Report, Harrisburg, PA, n.p

Skousen JG, Rose AW, Geidel G, Foreman J, Evans R, Hellier W (1998) Handbook of technologies for avoidance and remediation of acid mine drainage. National Mine Land Reclamation Center, Morgantown, WV, USA, 131 pp

Steiner L (2000) Pennsylvania fishes. Pennsylvania Fish and Boat Commission, Harrisburg, PA, 170 pp

Stuart WT, Schneider WJ, Crooks JW (1967) Swatara Creek Basin of southeastern Pennsylvania--An evaluation of its hydrologic system. US Geol Surv Water-Supply Paper 1829, 79 pp

Szpir L, Spooner J, Line DE, Meals D, Grabow GL, Osmond DL (2007) Section 319 National Monitoring Program Projects, National Nonpoint Source Watershed Project Studies, 2007 Summary Report. North Carolina State University Biological and Agricultural Engineering Department, Water Quality Group, p. 275-286. (http://www.bae.ncsu.edu/programs/extension/wqg/319monitoring/07rept319/index.htm)

Tomkiewicz SM, Dunson WA (1977) Aquatic insect diversity and biomass in a stream marginally polluted by acid strip mine drainage. Water Research 11:397-402

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

U.S. Environmental Protection Agency (1995) 1995 progress report--statement of mutual intent strategic plan for the restoration and protection of streams and watersheds polluted by acid mine drainage from abandoned coal mines. US EPA, Philadelphia, PA, 33 pp

U.S. Environmental Protection Agency (2002) National recommended water quality criteria--2002. US EPA 822-R-02-047, 33 pp

U.S. Environmental Protection Agency (2007) Section 319 nonpoint source success stories--Pennsylvania: Swatara Creek. US EPA 841-F-07-001P, 2 pp. (http://www.epa.gov/owow/nps/Success319/state/ pa_swatara.htm)

U.S. Geological Survey (2000) Land cover. US Geol Surv Map 66, scale 1:500,000

US Geological Survey (variously dated) USGS annual hydrologic data report of Pennsylvania--Swatara Creek Basin and Swatara Creek project. US Geol Surv Pennsylvania Water Science Center (http:// pa.water.usgs.gov/ar/index.html)

Vrba J, Kopacek J, Fott J, Kohout L., Nedbalova L, Prazakova M, Soldan T, Schaumburg J (2003) Long- term studies (1871–2000) on acidification and recovery of lakes in the Bohemian Forest (central Europe). Sci Total Environ 310:73–85

Wagner RJ, Mattraw HC, Ritz GF, Smith BA (2000) Guidelines and standard procedures for continuous water-quality monitors: site selection, field operation, calibration, record computation, and reporting. US Geol Surv Water-Resour Inv Rep 00-4252, 53 pp

Watzlaf GR, Schroeder KT, Kleinmann RLP, Kairies CL, Nairn RW (2004) The passive treatment of coal mine drainage. US DOE/NETL-2004/1202, 72 pp

Webster JG, Swedlund PJ, Webster KS (1998) Trace metal adsorption onto an acid mine drainage iron(III) oxy hydroxy sulfate. Environ Sci Tech 32:1361-1368

Winland RL, Traina SJ, Bigham JM (1991) Chemical composition of ochreous precipitates from Ohio coal mine drainage: Jour Environ Qual 20:452-460

Wolkersdorfer C, Bowell RJ (Eds) (2004) Contemporary reviews of mine water studies in Europe, part 1. Mine Water Environ 23:162-182

Wolkersdorfer C, Bowell RJ (Eds) (2005a) Contemporary reviews of mine water studies in Europe, part 2. Mine Water Environ 24:2-37

Wolkersdorfer C, Bowell RJ (Eds) (2005b) Contemporary reviews of mine water studies in Europe, part 3. Mine Water Environ 24:58-76

Wood CR (1996) Water quality of large discharges from mines in the anthracite region of eastern Pennsylvania. US Geol Surv Water-Resour Inv Rep 95-4243, 69 pp

Wood GH Jr, Kehn TM, Eggleston JR (1986) Deposition and structural history of the Pennsylvania Anthracite region. In: Lyons, PC, Rice, CL, editors. Paleoenvironmental and Tectonic Controls in Coal- Forming Basins of the United States. Geol Soc America Special Paper 210:31-47

Wood GH Jr, Trexler JP, Kehn TM (1968) Geologic maps of anthracite-bearing rocks in the west-central part of the southern Anthracite Field Pennsylvania, eastern area. US Geol Surv Miscellaneous Geologic Investigations Map I-528

Yan ND, Leung B, Keller W, Arnott SE, Gunn JM, Raddum GG (2003) Developing conceptual frameworks for the recovery of aquatic biota from acidification. Ambio 32:165-169

Youndt JD, Niemi GJ (1990) Recovery of lotic communities and ecosystems from disturbance--A narrative review of case studies. Environ Management 14:547-569

Ziemkiewicz PF, Skousen JG, Brant DL, Sterner PL, Lovett RJ (1997) Acid mine drainage treatment with armored limestone in open limestone channels. J Environ Qual 26:1017-1024

Ziemkiewicz PF, Skousen JG, Simmons J (2003) Long-term performance of passive acid mine drainage treatment systems. Mine Water Environ 22:118-129

PUBLISHED REPORTS/BOOK CHAPTERS/JOURNAL ARTICLES

Cravotta, C.A., III, 1998, Oxic limestone drains for treatment of dilute, acidic mine drainage, in Proceedings 19th Annual Meeting West Virginia Surface Mine Drainage Task Force: Morgantown, W.Va., West Virginia University, 25 p. (http://mine-drainage.usgs.gov/pubs/cravotta/WV98cra.htm)

Cravotta, C.A., III, 2000, Relations among sulfate, metals, sediment, and streamflow data for a stream draining a coal-mined watershed in east-central Pennsylvania, in Proceedings Fifth International Conference on Acid Rock Drainage: Littleton, Co., Society for Mining, Metallurgy, and Exploration, Inc., v. 1, p. 401-410. (http://mine-drainage.usgs.gov/pubs/cravotta/ICARD.pdf).

Cravotta, C.A., III, 2002, New method to estimate longevity of anoxic limestone drains (poster), in Proceedings of the 19th Annual Meeting of American Society of Mining & Reclamation, Lexington, Kentucky, June 9-13, 2002: American Society of Mining & Reclamation, p. 1062-1064.
(http://mine-drainage.usgs.gov/pubs/cravotta/ald_poster.pdf).

Cravotta, C.A., III, 2003, Size and performance of anoxic limestone drains to neutralize acidic mine drainage: Journal of Environmental Quality, v. 32, p. 1277-1289. http://mine-drainage.usgs.gov/pubs/ cravotta/preprint_q02-0041.pdf)

Cravotta, C.A., III, [2009], Abandoned mine drainage in the Swatara Creek Basin, Southern Anthracite Coalfield, Pennsylvania, USA--1. Streamwater-quality trends coinciding with the return of fish.: Mine Water and the Environment.

Cravotta, C.A., III, [2009], Abandoned mine drainage in the Swatara Creek Basin, Southern Anthracite Coalfield, Pennsylvania, USA--2. Performance of passive-treatment systems: Mine Water and the Environment.

Cravotta, C.A., III, and Bilger, M.D., 2001, Water-quality trends for a coal-mined watershed in Eastern Pennsylvania: Geochemistry, Exploration, Environment, Analysis, v. 1, p. 33-50. (http://mine-drainage.usgs.gov/pubs/cravotta/geaa_final.pdf).

Cravotta, C.A., III, Bilger, M.D., Brightbill, R.A., and Bogar, D., 2002, Restoration and monitoring of aquatic quality in a coal-mined watershed, Swatara Creek at Ravine, Pennsylvania (poster), in Proceedings of the 19th Annual Meeting of American Society of Mining & Reclamation, Lexington, Kentucky, June 9-13, 2002: American Society of Mining & Reclamation, p. 1061. (http://mine-drainage.usgs.gov/pubs/cravotta/fish_poster.pdf).

Cravotta, C.A., III, and Trahan, M.K., 1996, Limestone drains to increase pH and remove dissolved metals from an acidic coal-mine discharge in Pennsylvania, in Proceedings of the 13th Annual Meeting of the American Society for Surface Mining and Reclamation, Knoxville, TN, May 19-25, 1996, p. 836-840.

Cravotta, C.A., III, and Trahan, M.K., 1999, Limestone drains to increase pH and remove dissolved metals from acidic mine drainage: Applied Geochemistry, v. 14, no. 5, p 581-606. (http://mine-drainage.usgs.gov/pubs/cravotta/ag_final.pdf).

Cravotta, C.A., III, Ward, S.J., Koury, D.J., and Koch, R.D., 2004, Optimization of limestone drains for long-term treatment of acidic mine drainage, Swatara Creek Basin, Schuylkill County, PA in 2004 National Meeting of the American Society of Mining and Reclamation and the 25th West Virginia Surface Mine Drainage Task Force, April 18-24, 2004: American Society of Mine Reclamation, p. 366-411. (http://mine-drainage.usgs.gov/pubs/cravotta/ASMR_366-411.pdf).

Cravotta, C.A., III, and Watzlaf, G.R., 2002, Design and performance of limestone drains to increase pH and remove metals from acidic mine drainage, in Naftz, D.L., Morrison, S.J., Fuller, C.C., and Davis, J.A., eds., Handbook of groundwater remediation using permeable reactive barriers-- Application to radionuclides, trace metals, and nutrients: San Diego, Ca., Academic Press, p. 19- 66. (http://dx.doi.org/ 10.1016/B978-012513563-4/50006-2)

Cravotta, C.A., III, and Weitzel, J.B., 2001, Detecting change in water quality from implementation of limestone treatment systems in a coal-mined watershed, Pennsylvania, in Proceedings 8th National Nonpoint Source Monitoring Workshop: U.S. Environmental Protection Agency EPA/ 905-R-01-008. (http://mine-drainage.usgs.gov/pubs/cravotta/nmp00_camera.pdf).

Hornberger, R.J., Koury, D.J., Cravotta, C.A. III, Wood, T., and Brady, K.B.C., 2006, Characteristics and treatment of mine drainage in the Anthracite Region of Pennsylvania, in Proceedings of the Annuall Meeting of the West Virginia Mine Drainage Task Force: Morgantown, WV, West Virginia Mine Drainage Task Force, 20 p. (http://wvmdtaskforce.com/proceedings/06/Hornbergerpres.pdf)

Robbins, E.I., Cravotta, C.A., III, Savela, C.E., and Nord, G.L., Jr., 1999, Hydrobiogeochemical interactions in “anoxic” limestone drains for neutralization of acidic mine drainage: Fuel, v. 78, p. 259-270. (http://dx.doi.org/10.1016/S0016-2361(98)00147-1)

Robbins, E.I., Cravotta, C.A., III, Savela, C.E., Nord, G.L., Jr., Balciauskas, K.A., and Belkin, H.E., 1997, Hydrobiogeochemical interactions on calcite and gypsum in "anoxic" limestone drains in West Virginia and Pennsylvania: University of Kentucky, Lexington, Ky., 1997 International Ash Utilization Symposium, p. 546-559. (http://energy.er.usgs.gov/products/Papers/IAUS_97/).

Williams, D.J., Bigham, J.M., Cravotta, C.A., III, Traina, S.J., Anderson, J.E., and Lyon, G., 2002, Assessing mine drainage pH from the color and spectral reflectance of chemical precipitates: Applied Geochemistry, v. 17, p. 1273-1286. (http://mine-drainage.usgs.gov/pubs/cravotta/ag_williams.pdf).

ABSTRACTS

Cravotta, C.A., III, 1997, Hydrogeochemical considerations for ground-water monitoring and remediation at surface coal mines (abs.): Proceedings 4th National Nonpoint-Source Watersheds Projects Workshop, p. 87-88.

Cravotta, C.A., III, 1999, Oxic limestone drains to increase pH and remove dissolved metals from acidic coal mine drainage, Pennsylvania (abs.), in National Meeting of the American Geophysical Union, San Francisco, California, December 1999: EOS Transactions, American Geophysical Union, v. 80, no. 46, abs. H22F03.

Cravotta, C.A., III, 1999, Trends and relations between concentrations of suspended sediment and metals for coal-mined watershed, Pennsylvania (abs.), in National Meeting of the Geological Society of America, Denver, Colorado, October 1999: Geological Society of America, GSA Abstracts with Programs, v. 13, no. 7, abs. 25077.

Cravotta, C.A., III, 1999, Oxic limestone drains for treatment of dilute, acidic discharges (abs.), in National Meeting of the National Association Abandoned Mine Land Programs, Seven Springs Mountain Resort, Pennsylvania, August 22-25, 1999: National Association of Abandoned Mine Land Programs.

Cravotta, C.A., III, 2004, Optimization of limestone drains for long-term treatment of acidic mine drainage, Swatara Creek Basin, Schuylkill County, PA (abs.) in Abandoned Mine Reclamation Conference, Indiana, Pennsylvania, June 9-12, 2004: Western Pennsylvania Coalition for Abandoned Mine Reclamation.

Cravotta, C.A., III, 2005, Laboratory and field evaluation of limestone dissolution in passive systems for neutralization of acidic mine drainage (abs.): Mine Water Treatment Technology Conference, Pittsburgh, PA August 15-18, 2005.

Cravotta, C.A., III, 2005, Field and laboratory evaluation of limestone dissolution in passive systems for neutralization of acidic mine drainage (abs.), in National Meeting of the Geological Society of America, Salt Lake City, Utah, October, 2005: Geological Society of America, GSA Abstracts with Programs, abs. 42-8.

Cravotta, C.A., III, 2005, Swatara Creek National Monitoring Project--Lessons from monitoring of coal-mine-drainage treatment systems and downstream reaches (abs.): 13th National Nonpoint Source Monitoring Workshop From Projects to Programs: Enhancing States' NPS Management Programs through Lessons Learned from NPS Monitoring Projects September 18-22, 2005 Raleigh, North Carolina.

Cravotta, C.A., III, and Bilger, M.D., 2001, Water-quality trends for a stream draining the southern Anthracite Field, Pennsylvania (abs.), in U.S. Geological Survey Appalachian Region Integrated Science Workshop Proceedings, Gatlinburg, Tennessee, October 22-26, 2001: U.S. Geological Survey Open-File Report 01-406, p. 21. (http://www.appalachianregionscience.usgs.gov/appal/ pdf/OFR01406part1.pdf).

Cravotta, C.A., III, Bilger, M.D., Brightbill, R.A., and Bogar, D., 2002, Restoration and monitoring of aquatic quality in a coal-mined watershed (abs), in Annual Meeting of the American Water Resources Association, Philadelphia, Pennsylvania, November 3-7, 2002: Falls Church, Va., American Water Resources Association, P-52.