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Abandoned Mine Drainage in the Swatara Creek Basin, Southern Anthracite Coalfield, Pennsylvania, USA: 2. Performance of Treatment Systems

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TECHNICAL ARTICLE

Abandoned Mine Drainage in the Swatara Creek Basin, Southern Anthracite Coalfield, Pennsylvania, USA: 2. Performance of Treatment Systems

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Abstract A variety of passive and semi-passive treatment systems were constructed by state and local agencies 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, the US Geological Survey collected water-quality data at upstream and downstream locations relative to each system eight or more times annually for a minimum of 3 years at each site 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 (ALD), 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 tonne of acid treated over an assumed service life of 20 years was computed. On the basis of these performance metrics, the limestone sand, ALD, 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 ALD 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 storm flow conditions.

Keywords Coal mines · Diversion well · Limestone sand · Limestone channel · Limestone drain · Wetland

Introduction

Acidic mine drainage (AMD) commonly is treated near the point of origin to neutralize acidity and remove dissolved and suspended pollutants before it reaches a stream. 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 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 treatment systems 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 stream water may be effective to meet downstream aquatic-quality goals.

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Various passive and semi-passive treatment systems have different advantages and disadvantages, and 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 'typical' flow and water-quality conditions (Skousen et al. 1998; Ziemkiewicz et al. 2003), though treatment effectiveness and downstream benefits may diminish as conditions deviate from normal. For example, the performance of a treatment system could decline with increased flow rate during runoff events because of decreased retention time and/or increased contaminant load. However, treatment performance for a wide range of flow conditions is poorly documented in the literature.

This paper evaluates the effectiveness of passive and semi-passive treatment systems for neutralizing acidity and removing metals and other pollutants from AMD and affected stream water in the upper Swatara Creek Basin in eastern Pennsylvania. Data collected by the US Geological Survey (USGS) over a wide range of flow conditions during June 1996 through June 2007 at AMD treatment sites within and immediately downstream of the mined area above Ravine, Pa. (Fig. 1), were used for this evaluation. A companion paper (Cravotta et al. 2010) evaluates the downstream water-quality trends and recovery of fish populations within affected stream reaches during the study period.

Description of Treatment Systems

During 1995–2008, various passive and semi-passive treatment systems were installed at selected locations to neutralize the AMD or the stream water at downstream sites (Figs. 1, 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), US Department of Energy (USDOE), and USGS.

Limestone-sand dosing and open limestone channels (Fig. 2a) are relatively simple passive treatment systems where limestone is added once or infrequently to the streambed or AMD discharge channel (Skousen et al. 1998; Ziemkiewicz et al. 1997). Generally, these treatment methods were selected to add alkalinity downstream of inaccessible or diffuse AMD sources that were mildly acidic and had relatively low concentrations of dissolved Al and Fe (<2 mg/L). Although average residence time in the treated reach would be less than 5 min, particles transported downstream could continue to dissolve. Forty tonnes (t) of limestone sand (<0.5 cm diameter), which can

Fig. 1 Locations of water-quality and streamflow monitoring sites relative to major AMD sources and associated treatment systems in the upper Swatara Creek Basin within the Southern Anthracite Coalfield, above Ravine (See Cravotta et al. 2010, Fig. 1a for location of map area). Year of implementation of treatment is indicated in parentheses. Monitoring site identification number (for example, E3-1) is indicated only for sites discussed in this paper. Monitoring sites are listed in Table 1
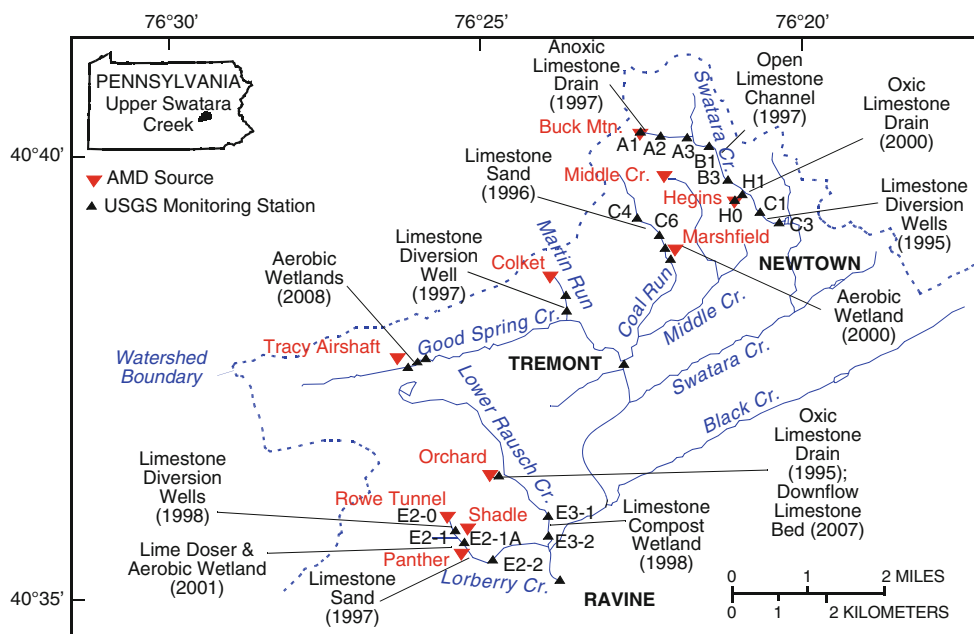
Fig. 2 Schematic illustrations of passive treatment systems installed in the upper Swatara Creek Basin, 1995–2008: **a** open limestone channel; **b** limestone diversion well; **c** anoxic or oxic limestone drain; and **d** limestone or compost based wetland. Although generalized, the illustration for **a** applies specifically to the upper Swatara Creek, that for **c** to treatment on the Buck Mountain discharge, and that for **d** to treatment on Lower Rausch Creek
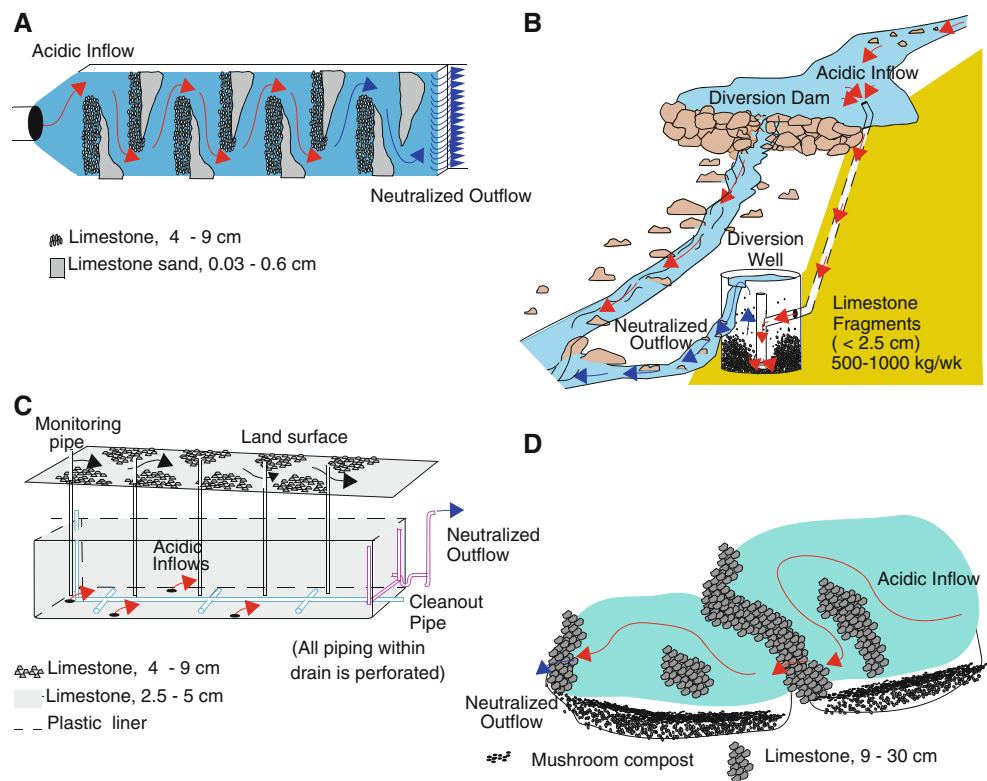
Table 1 Acidic mine drainage treatment systems and associated water-quality monitoring sites, upper Swatara Creek Basin, Schuylkill County, Pennsylvania; treatment system and monitoring site locations are shown in Fig. 1

Treatment system	Description	Year of installation/expansion	Monitoring site ID	
			Upstream	Downstream
LSC	Limestone sand in Coal Run below Middle Creek discharges	1996	C4	C6
OLS	Open limestone channel on Swatara Cr below Buck Mountain discharge	1997	B1	B3
ODH	Oxic limestone drain on Hegins discharge	2000, 2005	H0	H1
ADB	Anoxic limestone drain on Buck Mountain discharge	1997, 2001, 2005	A1	A2 and A3
DWS	Limestone diversion wells on Swatara Creek below Hegins discharge	1995	C1	C3
DWL	Limestone diversion wells on Lorberry Creek below Rowe Tunnel discharge	1998	E2-0	E2-1
WLL	Aerobic wetlands beside Lorberry Creek below Rowe Tunnel discharge	2001	E2-1A	E2-2
WLR	Limestone-compost wetlands on Lower Rausch Cr below Orchard discharge	1998	E3-1	E3-2

dissolve rapidly because of its small grain size and surface area, were dumped once from trucks directly into Coal Run downstream from the Middle Creek discharges, between sites C4 and C6, in September 1996, and 136 t were dumped into an unnamed tributary of Lorberry Creek below the Pantherhead discharge in February 1997 (Fig. 1). An open limestone channel was constructed in March 1997 within a 33.5 m long segment of Swatara Creek, between sites B1 and B3 (Fig. 1), below the confluence of acidic headwaters of Swatara Creek and the unnamed tributary that originated at the Buck Mountain discharge. To construct the open limestone channel, a total of 40 t of sand-size limestone fragments and 63 t of cobble-size 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 (Fig. 2a).

An anoxic limestone drain (ALD) (Fig. 2c) is another relatively simple passive-treatment method that involves the burial of cobble-size limestone aggregate in trenches that intercept acidic water before it emerges from the ground (Cravotta and Trahan 1999; Hedin et al. 1994a, b; Skousen et al. 1998). Generally, alkalinity production in

enclosed limestone drains is greater than that for open limestone systems (Cravotta 2003). Thus, if concentrations of dissolved Al and Fe^{3+} were low to moderate (<5 mg/L) and if space was accessible to install the limestone bed at the AMD source, ALDs were the preferred treatment method in the Swatara Creek watershed. Limestone drains designed for a typical retention time of 6 h were constructed in March 1995 at the Orchard discharge to treat a small oxic discharge (38–113 L/s; 40 t of limestone) along Lower Rausch Creek; in May 1997 at the Buck Mountain discharge (site A1) to treat a large, anoxic discharge (189–756 L/s; 320 t of limestone) at the headwaters of Swatara Creek; and in June 2000 at the Hegins discharge (site H0) to treat a large oxic discharge (378–1,890 L/s; 727 t of limestone) near the headwaters of Swatara Creek (Figs. 1, 2).

The limestone beds at the Orchard and Buck Mountain discharges were buried to minimize exchange with the atmosphere. Keeping 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). Perforated flushing pipes were installed within the limestone bed at the Buck Mountain and Hegins discharges to facilitate the removal of precipitated Fe^{III} and Al oxyhydroxides.

Because monitoring indicated substantial limestone dissolution, the ALD on the Buck Mountain discharge was supplemented twice, in January 2001 and September 2005, with 91 t of limestone. Additionally, in September 2005, the oxic limestone drain on the Hegins discharge was enlarged with the addition of 182 t of limestone and covered with approximately 0.15 m of leaf-litter compost. The enlargement and compost cover were intended to increase retention time, retain CO_2, and promote greater rates of limestone dissolution. Lastly, in August 2007, the oxic limestone drain on the Orchard discharge, which had been out of service since 2000, was completely reconstructed as an up-flow 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 treatment system is not described further.

In a limestone diversion well (Fig. 2b), acidic stream water or 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). Generally, if the AMD source was net acidic with moderate or elevated concentrations of dissolved O_2, Al, or Fe^{3+} (>2 mg/L) and if space was limited for construction of a treatment system, limestone diversion wells were installed at an accessible downstream location. As much as 1 t 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; in July 1997, a single diversion well was installed to treat water downstream from the Colket discharge 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 discharge below site E2-0 near the headwaters of Lorberry Creek (Fig. 1). Because the Martin Run diversion well clogged repeatedly and was rarely working during the subject investigation, this treatment system is not described further.

Constructed wetlands or settling ponds are a typical component of most AMD treatment systems that promote 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 (Fig. 2d) 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 reduce the downstream transport of suspended metal-rich particles in the upper Swatara Creek Basin. In December 1997, near the mouth of Lower Rausch Creek at site E3 (Fig. 1), a 0.93-ha limestone-compost-based wetland was constructed to remove metals from stream flow that commonly had near-neutral pH but had potential to be net acidic during storm flow 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 (Fig. 1, site E3-S0). Additionally, in December 2001, a 0.49-ha wetland was constructed adjacent to Lorberry Creek at station E2-1 (Fig. 1). 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. 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, these treatment systems are not described further.

Methods

To document variations in untreated AMD, treatment-system performance, and cumulative downstream effects of AMD treatment monitoring sites were established upstream and downstream of each treatment and along lower reaches of Swatara Creek (Fig. 1). Fixed-interval grab samples (4-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. 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 (American Public Health Association 1998a) within 24 h of sampling, whereas hot-peroxide acidity (American Public Health Association 1998b), total constituent concentrations, and 'dissolved' (0.45 μm pore-size filter) constituent concentrations were analyzed within 3 months of sampling. Because hot-peroxide acidity values obtained for this study did not include results for negative values, the net acidity was computed considering positive contributions from H^+ (pH) and concentrations of dissolved iron, manganese, and aluminum in mg/L (C_{Fe}, C_{Mn}, C_{Al}, respectively) as:

$$\text{Net Acidity(mg/L CaCO}_3) = 50\left(10^{(3-\text{pH})}\right)$$
$$+ 2C_{Fe}^{2+}/55.85 + 2C_{Mn}/54.94 + 3C_{Al}/26.98$$
$$- \text{Alkalinity} \qquad (1)$$

Kirby and Cravotta (2005) showed that net acidity computed with Eq. 1 is comparable in value to the standard method hot peroxide acidity (American Public Health Association 1998b). They also showed that if the AMD is net acidic (net acidity > 0; hot-peroxide acidity > 0), the ultimate pH of oxidized samples will be less than 5.0;

however, if the AMD is net alkaline (net acidity < 0; hot-peroxide acidity < 0), the ultimate pH of the oxidized AMD will be maintained at values greater than or equal to 6.0.

The computed net acidity and associated hydrochemical data for influent and effluent samples or upstream and downstream samples for eight individual treatment systems (Table 1) were compared to evaluate performance. 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 and water quality 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 for AMD sources 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 stream flow of Swatara Creek at Ravine on the date of sampling was less than the 25th percentile for the study period, the sample was classified as low-flow; between the 25th and 75th percentiles, the sample was classified as normal-flow; or greater than the 75th percentile, the sample was classified as 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 used by Ziemkiewicz et al. (2003) for 83 different treatment systems in the eastern US, the acid-removal efficiency was computed as the median acid load removed (influent net-acidity load–effluent net-acidity load, in g/day as CaCO₃) divided by the size of the treatment system. The size of wetland systems was indicated by the total surface area (in m²) of ponds and wetlands; in contrast, the size of limestone systems was indicated by the total mass of limestone

(in t) installed during the elapsed years in service. The mass of limestone for diversion wells was estimated as 30 t 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 tonne 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.

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 loads (Growitz et al. 1985). The major AMD sources studied during the previous and current investigations had high contaminant loads associated with high flow rates (medians greater than 100 L/min), such as the Tracy Airshaft, Rowe Tunnel, Middle Creek, Colket, Buck Mountain, and Hegins discharges, or low flow rates with elevated concentrations of dissolved metals, such as the Pantherhead, Shadle, and Orchard discharges (Fig. 3; Table 2). Depending on the AMD source, the flow rate at a given site varied by 1–3 log units during the current study; associated chemical variations were less pronounced. The larger volume AMD sources generally had the least-variable flow rates and chemistry. Although the Tracy Airshaft and Marshfield discharges were consistently net alkaline, the other large discharges were net acidic with near-neutral pH (>5) and elevated concentrations of dissolved Fe (>3 mg/L). The smaller volume AMD sources were net acidic with low pH (<4.5) and elevated concentrations of Fe, Al, Ni, and Zn (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 and DO (Fig. 3), generally indicate redox-controlled, kinetic limitations on the precipitation of oxidized compounds of these metals (e.g. Cravotta 2008a). Likewise decreased concentrations of dissolved Al with increased pH are consistent with solubility control by Al-hydroxide (e.g. Cravotta 2008a).

During the current study (1996–2007), the Shadle discharge exhibited the widest variability in water quality compared to other AMD sources in the watershed (Fig. 3). The net acidity of the Shadle discharge decreased progressively from a median value of 1100 mg/L $CaCO_3$ for 1996–1998 to a median value of 180 mg/L $CaCO_3$ for 2005–2007, while pH increased progressively from values of 3.1–3.2 in 1996–1998 to values of 4.9–6.2 in 2005–2007. 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 instead from the rapid flooding of this underground mine following its closure around 1990. Permanent flooding of a mine can result in: (1) dissolution of accumulated pyrite oxidation products; (2) a decrease in the amount of oxygen that reaches 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 apparent natural improvement in AMD and surface-water quality and 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 1996–1998, streamwater of Swatara Creek at Newtown and Ravine (Fig. 1) ranged from mildly acidic to near-neutral (net acidity -20 to 10 mg/L $CaCO_3$; pH 4.5–8.0) with moderate concentrations of dissolved solids (SC 60–400 μS/cm) that varied as a function of stream flow (Cravotta and Bilger 2001; Cravotta and Weitzel 2001). Higher values of pH, SC, and SO_4 were associated with base-flow conditions sustained by near-neutral groundwater and net-alkaline AMD in the upper part of the watershed, such as the Tracy Airshaft and Colket discharges (Fig. 3). Lower values of pH, SC, and SO_4 were associated with acidic storm runoff (Cravotta et al. 2010).

In contrast with Swatara Creek at Newtown and Ravine, the pH and SC for the headwaters of Lorberry Creek, below the Rowe Drainage Tunnel (Fig. 1), 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 acidic water to the overlying mine pool. When the pumping was active, net acidity of Lorberry
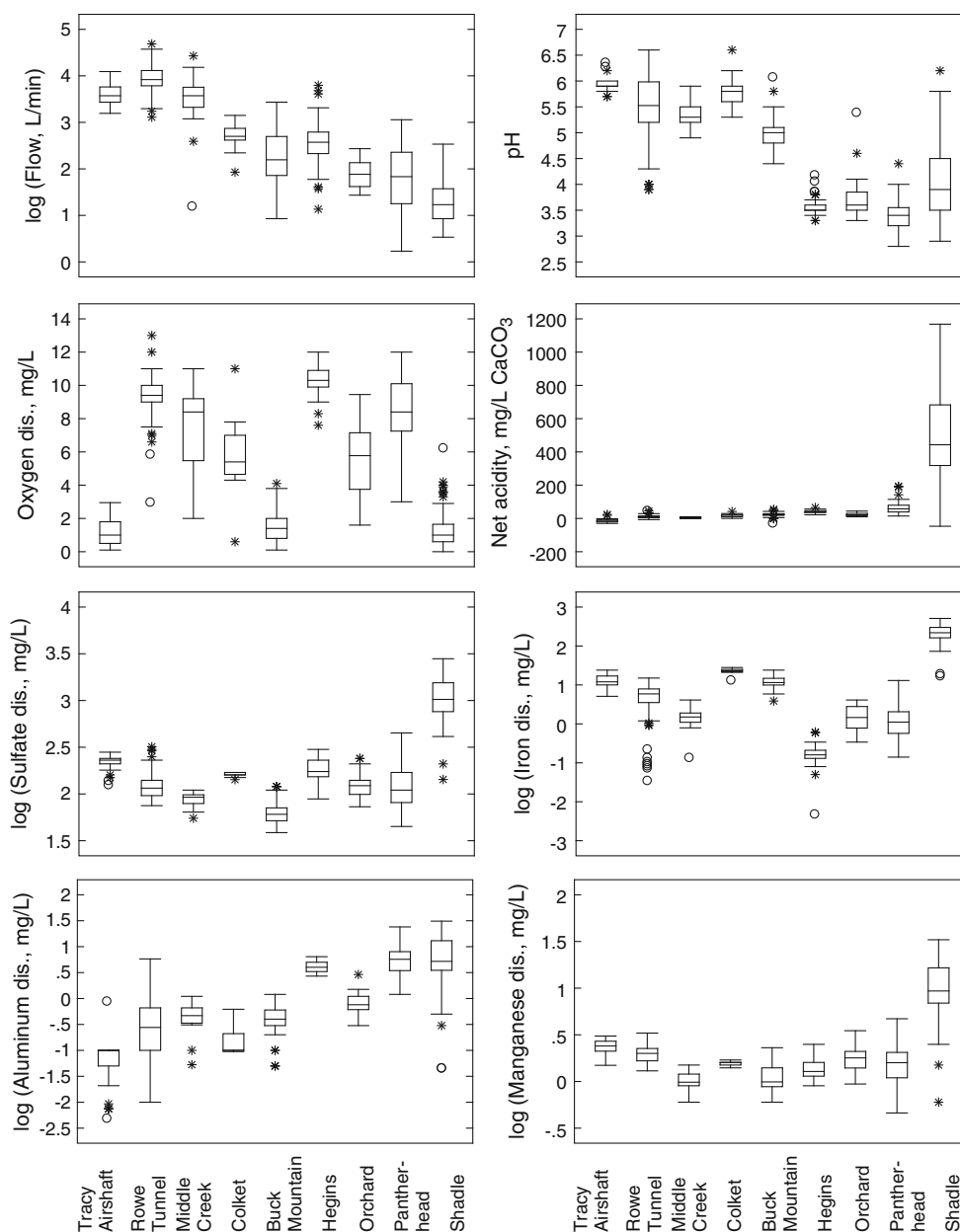
Fig. 3 Boxplots summarizing hydrochemical characteristics of AMD sources upstream from any treatment in the Swatara Creek Basin, Pa, 1996–2007. Area of *box* indicates the interquartile range (IQR = 25th to 75th percentile); *horizontal line* inside the *box* indicates the median; *vertical lines* extend to extreme values within 1.5 times the IQR; * and o *symbols* indicate outlier values that are greater than 1.5 but less than 3 times the IQR outside the quartile, and greater than three times the IQR outside the quartile, respectively

Creek increased by 10–20 mg/L $CaCO_3$, pH declined by 0.5–1 unit, and SC increased by 50–200 μS/cm. These short-term fluctuations in pH and SC were apparent in the continuous monitoring data for 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 of Lorberry Creek, treatment systems were implemented along stream reaches

downstream from the AMD where access and space were not limiting (Fig. 1; Table 1).

Evaluation of Treatment Performance

Generally, all eight of the treatment systems evaluated removed acidity, as indicated by significant downstream decreases (matched pair tests) in the net acidity concentration and load (Figs. 4, 5, 6). However, the median acid

Table 2 Median water quality and constituent loading for AMD in upper Swatara Creek Basin, 1996–2007

Constituent	AMD sites									
	Tracy airshaft	Rowe tunnel	Middle Creek	Marsh-field[a]	Colket	Buck Mtn	Hegins	Orchard	Panther-head	Shadle
No 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	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 the Marshfield discharge before a wetland constructed in 2000 flooded the site, preventing access

load removed (Table 3) and the magnitude of effects, if any, on the flow rate and other water-quality constituents varied widely among the treatment systems. Although upstream and downstream time-series data for flow rates and a variety of water-quality constituents were evaluated and are accessible in an electronic appendix net-acidity data are emphasized in the following evaluations of the treatment performance.

For example, as implied by decreased acidity, the limestone drains at the Hegins and Buck Mountain discharges (ODH, ADB) and the limestone diversion wells on Swatara Creek and Lorberry Creek (DWS, DWL) increased the pH and decreased the dissolved Fe and Al loads 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 (Fig. 6). 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, whereby several truckloads of finely crushed limestone were spilled at once over the stream bank into the channel. As the mound of limestone sand was eroded at the base, fresh limestone spilled into the channel where it gradually dissolved. Stream flow in the treated section of Coal Run ranged from 8.5 to 215 L/s, and originated as AMD from several sources similar in quality to the Middle Creek and Marshfield discharges (Fig. 3; Table 2). The stream water above (C4) and below (C6) the treated reach had similar flow and water quality, characterized by net
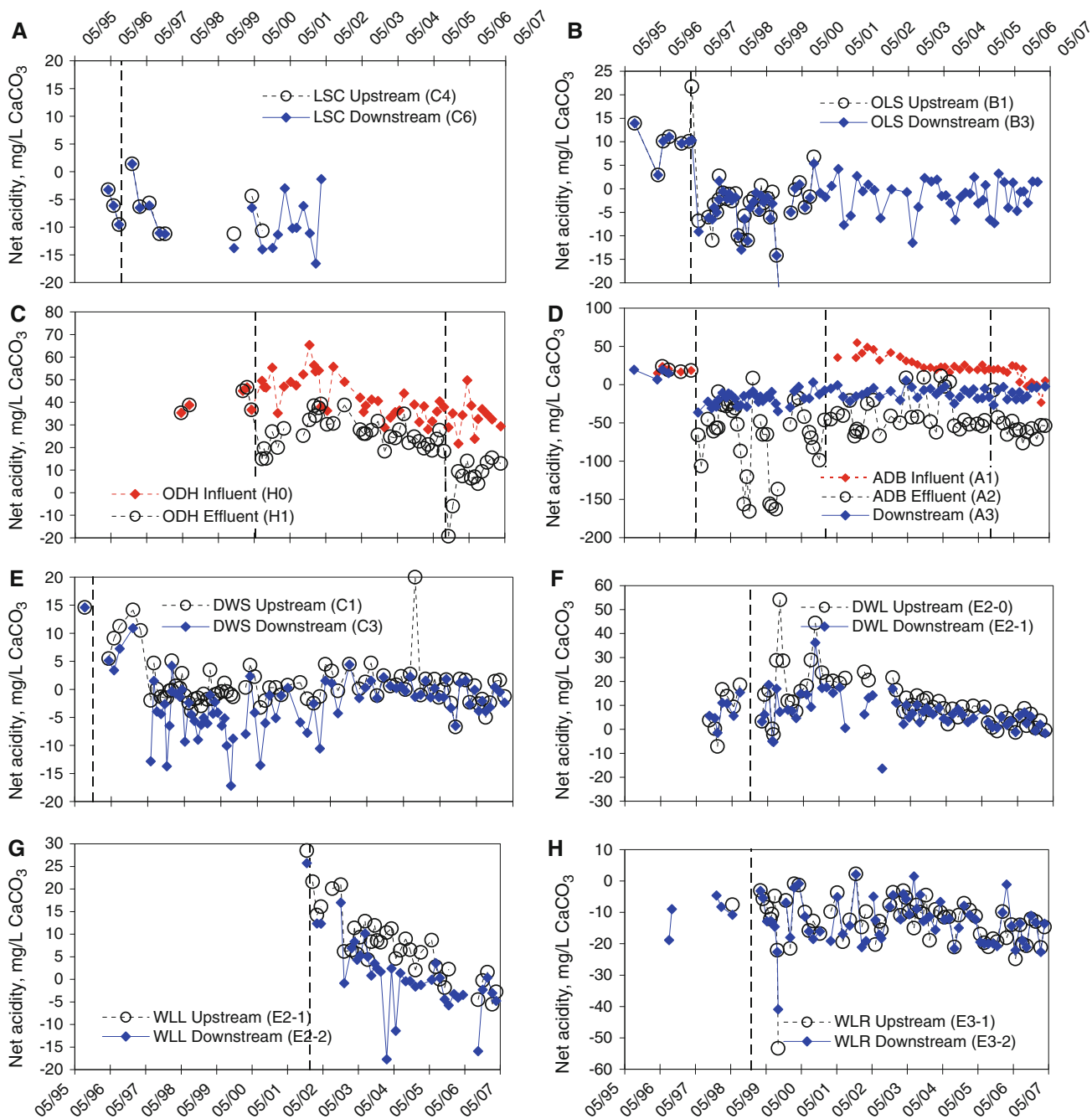
Fig. 4 Net acidity concentrations upstream and downstream of selected treatment systems in the Swatara Creek Basin; *vertical dashed lines* indicate when treatment started or supplemental addition of limestone: **a** limestone sand in Coal Run (LSC); **b** open limestone channel on Swatara Creek (OLS); **c** oxic limestone drain at Hegins discharge (ODH); **d** anoxic limestone drain at Buck Mountain discharge (ADB); **e** limestone diversion wells on Swatara Creek (DWS); **f** limestone diversion wells on Lorberry Creek (DWL); **g** aerobic wetlands on Lorberry Creek (WLL); **h** aerobic limestone-compost wetlands on Lower Rausch Creek (WLR)

acidity concentrations from 11.2 to 1.4 mg/L $CaCO_3$ (Fig. 4a), pH values from 5.6 to 6.9, and moderate concentrations of dissolved metals (Fe 0.5–2.0 mg/L; Mn 0.76–1.2 mg/L; Al < 0.5 mg/L) (Supplementary material). Although the matched-pair tests indicated the downstream concentrations and loads of net acidity decreased and of dissolved Ca and Mn increased (Fig. 6), the pH and dissolved concentrations and loads of Fe and Al were not significantly different between the upstream and downstream sites for most conditions. Increased Mn was not expected, but could be an impurity in the limestone. The limestone-sand treatment on Coal Run removed a median
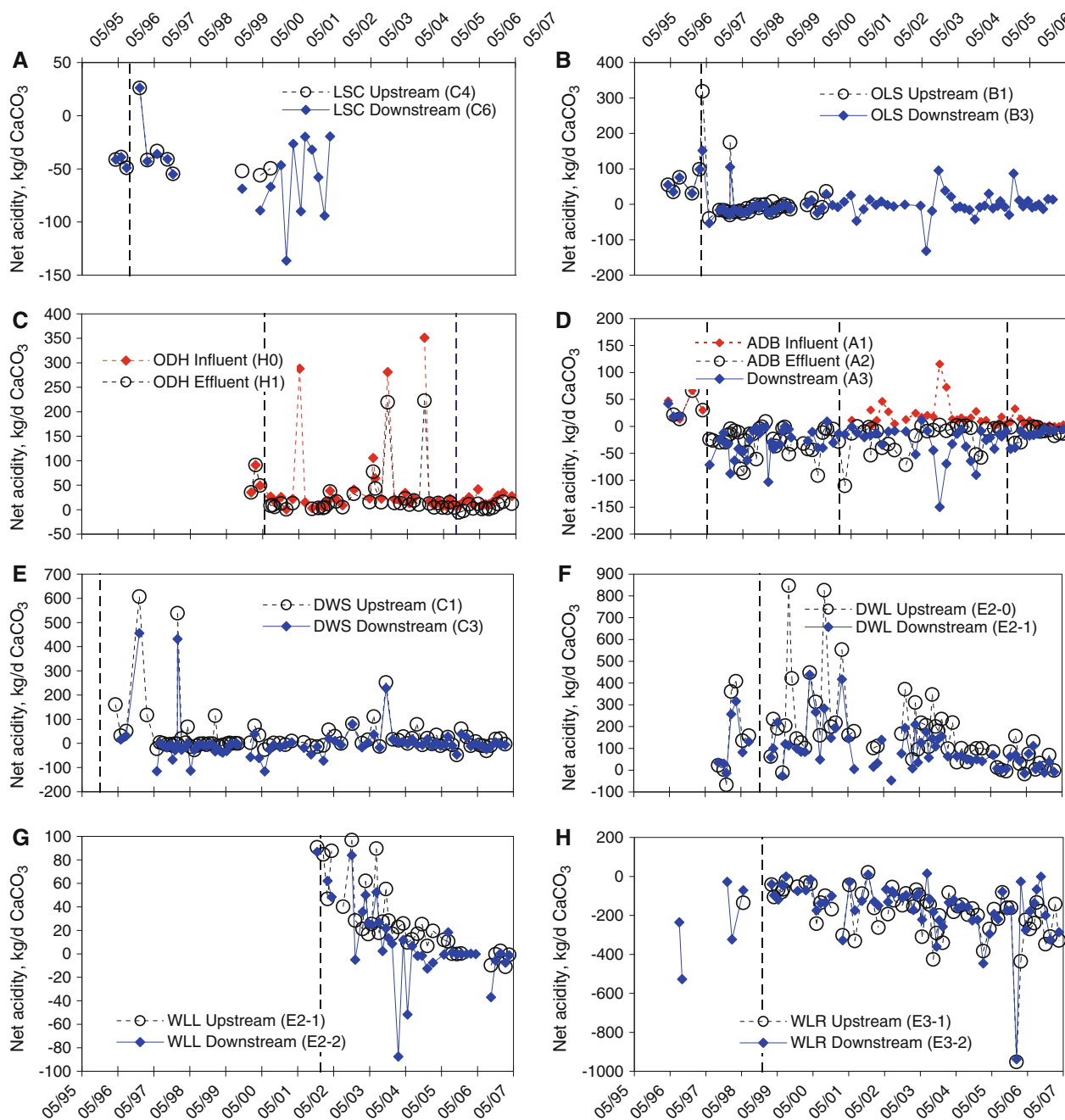
Fig. 5 Net acidity loads upstream and downstream of selected treatment systems in the Swatara Creek Basin; vertical dashed lines indicate when treatment started or supplemental addition of limestone: **a** limestone sand in Coal Run (LSC); **b** open limestone channel on Swatara Creek (OLS); **c** oxic limestone drain at Hegins discharge (ODH); **d** anoxic limestone drain at Buck Mountain discharge (ADB); **e** limestone diversion wells on Swatara Creek (DWS); **f** limestone diversion wells on Lorberry Creek (DWL); **g** aerobic wetlands on Lorberry Creek (WLL); **h** aerobic limestone-compost wetlands on Lower Rausch Creek (WLR)

acidity of 1.9 t/year and added 2.2 t/year Ca as $CaCO_3$ over the 6-year monitoring period (Figs. 5a, 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 estimated cost efficiency ($108/t) (Table 3).

Open Limestone Channel on Swatara Creek (OLS; B1–B3)

Before implementing treatment in March 1997, the stream water at sites B1 and B3 was acidic (3–14 mg/L $CaCO_3$) (Fig. 4b), with low pH (<4.5), low concentrations of SO_4
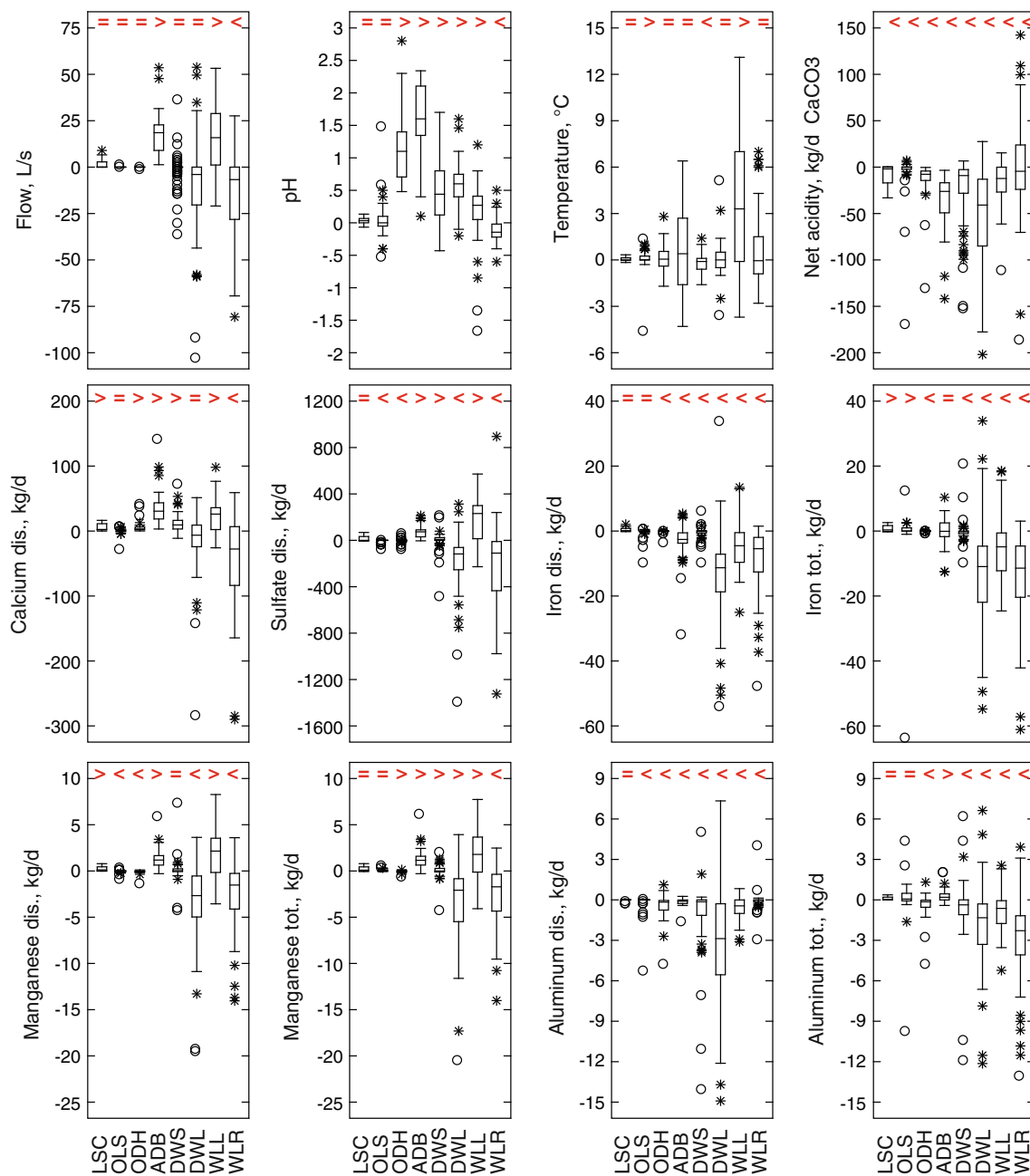
Fig. 6 Boxplots of post-implementation data showing differences in water quality between downstream and upstream sites for selected treatment systems in the Swatara Creek Basin. *Axis* labels are: LSC, limestone sand in Coal Run (C6–C4); OLS, open limestone channel on Swatara Creek (B3–B1); ODH, oxic 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 Fig. 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)

(12–48 mg/L) and Mn (<0.5 mg/L), and moderate concentrations of dissolved Al and Fe (0.5–3 mg/L) (Supplementary material). These conditions and preliminary field experiments indicating initially rapid increases in the pH and minor decreases in acidity of stream water in

contact with limestone in an 'open container' warranted the construction of the open limestone channel. The preliminary experiments indicated the rates of limestone dissolution and acid neutralization decreased with increased pH, which is consistent with open container

Table 3 Characteristics of treatment systems in the upper Swatara Creek Basin, including flow (L/s), median influent (Infl) and effluent (Effl) quality, acid load removed or $CaCO_3$ load added (t/year), size as mass (t) or area (m^2), years in service, estimated costs (US$), removal efficiency (R.E.) (g/d/t) or (g/d/m^2)[ab], and cost efficiency (C.E.) ($/t)[c] for

| | No of sample pairs | Flow | pH | | Net acidity (mg/L) | | Acid load removed | CaCO3 load added | Size[d] | Years in service | Cost[e] | R.E. | C.E. |
			Infl	Effl	Infl	Effl							
LSC	11	56.6	6.5	6.5	−8.5	−8.8	−1.9	2.2	40 t	11	1,500	47.5	108
OLS	39	62.3	6.4	6.5	−2.8	−3.0	−0.2	0.1	100 t	10	3,500	2.0	2,397
ODH	50	5.4	3.5	4.6	38.2	23.2	−7.7	3.0	909 t	7	50,000	8.5	891
ADB	38	24.6	5.0	6.6	23.2	−10.1	−26.2	27.9	502 t	10	25,000	52.1	131
DWS	82	82.4	5.5	6.0	0.0	−1.6	−9.1	8.6	840 t	12	68,800	12.7	1,035
DWL	64	131.7	5.5	6.2	9.3	6.2	−40.9	1.3	630 t	9	68,800	75.8	230
WLL	33	56.1	5.7	6.0	6.5	0.6	−12.3	23.8	4,860 m^2	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 m^2	9	175,000	0.3	7,783

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

[b] R.E. for limestone-based systems expressed as g of acidity as $CaCO_3$ removed per day per t of limestone; for wetlands, computed as acid load removed divided by area of wetland, multiplied by conversion factor of 1,000 g/kg

[c] C.E. expressed as $/t of acidity as $CaCO_3$ removed, assuming a 20 year service life, multiplied by a conversion factor of 2.74 (d·t)/(year·kg)

[d] Size of limestone-based system in tonnes of limestone; size of wetland in m^2

[e] The cost to build and maintain the diversion wells was estimated, assuming $20,000 for initial well installation with hopper storage, $24/t for limestone over 20 years, and $1,000 a year for operation and maintenance (filling wells, clearing debris from intakes)

testing of nearby AMD (e.g. Cravotta 2003; Cravotta et al. 2004).

The first pair of post-implementation samples collected in April 1997 indicated net acidity decreased by 11 mg/L $CaCO_3$ or 50% (Figs. 4b, 5b) and pH increased by 0.5 unit (Supplementary material). However, water quality benefits were not as pronounced after May 1997, following the installation of an anoxic limestone drain on the Buck Mountain discharge at site A2 that produced near-neutral stream water at site B1 above the open limestone channel. The neutral upstream water was not aggressive toward limestone in the stream channel compared to the initially acidic stream water. Nevertheless, the concentrations and loads of net acidity and dissolved SO_4, Al, and Mn exhibited significant decreases between the upstream site (B1) and downstream site (B3) at the open limestone channel (Fig. 6). Overall differences in pH and concentrations of dissolved Ca and Fe were not significant for any flow conditions (Fig. 6). The limestone channel treatment on Swatara Creek removed a median acidity of 0.2 t/year and added 0.1 t/year Ca as $CaCO_3$ over a 4-year monitoring period with paired samples (Fig. 6; 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 worst estimated cost efficiency ($2,397/ t) (Table 3). Considering initial results when the influent stream water was acidic, the efficiency may be expected to decrease as the influent quality improves.

Oxic Limestone Drain on Hegins Discharge (ODH; H0–H1)

Before implementing treatment in June 2000, untreated AMD flowed from the Hegins discharge at site H0 for about 100 m as an unnamed tributary to site H1 at Swatara Creek (Fig. 1). The oxic limestone drain was constructed as a series of limestone-filled cells within the channel of this unnamed tributary. The untreated AMD was oxic (DO 7.6–12 mg/L) and acidic (net acidity 22–65 mg/L) with low pH (3.3–4.2) and elevated concentrations of dissolved Al (2.7–6.4 mg/L) and Mn (0.9–2.5 mg/L) but moderate concentrations of dissolved Fe (<1 mg/L) (Fig. 3). Compared to the influent, the treated effluent had lower concentrations of net acidity (Fig. 4c), higher pH (4.0–6.9; median increase of 1.1), and lower concentrations of dissolved metals (Supplementary material). After the system was enlarged with additional limestone and covered with compost in September 2005, the treatment effectiveness improved (Fig. 4c and Supplementary material). Over the 7 year monitoring period, the oxic limestone drain on the Hegins discharge removed a median acidity of 7.7 t/year and added 3.0 t/year Ca as $CaCO_3$ (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 estimated cost ($891/t) (Table 3).

Anoxic Limestone Drain on Buck Mountain Discharge (ADB; A1–A3)

Before implementing treatment in May 1997, untreated 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 (Fig. 1). The untreated AMD generally was suboxic (DO < 2.0 mg/L) with slightly acidic pH (4.1–6.1) and elevated concentrations of dissolved Fe (4–24 mg/L) but moderate concentrations of dissolved Al (≤1.2 mg/L) and Mn (≤2.3 mg/L) (Fig. 3). 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) (Supplementary material).

After implementation of the ALD, the downstream water at site A3 had positive net alkalinity (net acidity < 0) (Fig. 4d) and consistently near-neutral pH (Supplementary material). In addition to the decreased acidity concentrations and loads (Figs. 4d, 5d), the effluent from the ALD at site A2 had significantly higher pH (median increase 1.6), greater concentrations and loads of Ca, and smaller concentrations and loads of dissolved Fe and Al than the influent at A1 (Fig. 6). These effluent characteristics were consistent with results of closed-container 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 (Supplementary material). 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 ALD was sufficient to buffer the downstream pH at site A3.

The ALD decreased the acidity by a median of 26.2 t/year and increased dissolved Ca by a median of 27.9 t/year 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 ALD, as explained in detail by Cravotta (2003) and Cravotta et al. (2004). Because of the rapid rate of limestone consumption, the Buck Mountain ALD, 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 ALD treatment had relatively high acid-removal efficiency (52.1 g/d/t) and a low estimated cost ($131/t) compared to the other treatment systems (Table 3).

The ALD 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 (Supplementary material). However, sporadically during normal to high-flow conditions, the concentration and load of total Al increased downstream (Supplementary material). During normal to high flows, turbulent water could transport freshly precipitated Al-hydroxide particles downstream.

Limestone Diversion Wells on Swatara Creek (DWS; C1–C3)

Before implementing treatment in July 1997, the stream water at sites C1 and C3 had the same acidity (Fig. 4e), pH < 4.5, dissolved Al > 1.5 mg/L, and associated constituents (Supplementary material). After implementation of the treatment, the stream water at the downstream site (C3), approximately 140 m below the diversion wells, typically had a concentration of acidity that was 2 mg/L $CaCO_3$ lower and a pH that was 0.5 unit higher than the upstream site (C1) (Fig. 6 and Supplementary material). The median decrease in net acidity load was 9.1 t/year, and the increase in Ca load was 8.6 t/year as $CaCO_3$. Assuming limestone consumption at a rate of 30 t/year for each diversion well, the treatment on Swatara Creek had relatively low acid-removal efficiency (12.7 g/d/t) and high estimated cost ($1,035/t) compared to the other treatment systems (Table 3).

For most flow conditions, the limestone diversion wells decreased concentrations of net acidity, increased pH and concentrations of Ca, and decreased dissolved Fe and Al (Fig. 6 and Supplementary material). 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 extreme storm flow conditions, the effectiveness of the limestone diversion wells was diminished because a smaller proportion of total stream flow was treated.

Limestone Diversion Wells on Lorberry Creek (DWL; E2-0–E2-1)

Lorberry Creek at site E2-0 consisted predominantly of effluent from the Rowe Drainage Tunnel, which had extremely variable net acidity −2.2 to 54 mg/L as $CaCO_3$) and pH (3.9–6.5) and elevated concentrations of dissolved Fe (2.0–12 mg/L) and Al (0.01–5.8 mg/L) (Figs. 3, 4f). Although the diversion wells on Lorberry Creek below Rowe Drainage Tunnel did not treat the entire flow of the Rowe Tunnel, they effectively decreased net acidity, increased pH, and decreased dissolved Fe and Al concentrations in the downstream segment of Lorberry Creek at

site E2-1 over a wide range of flow conditions (Fig. 4f and Supplementary material). The median decrease in net acidity load was 40.9 t/year, but the increase in Ca load was only 1.3 t/year as $CaCO_3$. Assuming limestone consumption at a rate of 30 t/year for each diversion well, the treatment on Lorberry Creek had very high acid-removal efficiency (75.8 g/d/t) and relatively low estimated cost ($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 soda ash (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 estimated cost 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 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.

Aerobic Wetlands Below Diversion Wells on Lorberry Creek (WLL; E2-1A–E2-3)

Before installation of the aerobic wetlands in December 2001 downstream from the diversion wells on Lorberry Creek, 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.

Compared to the influent, the effluent from the wetlands downstream of the diversion wells on Lorberry Creek had lower concentrations and loads of acidity and dissolved and total Fe and Al plus higher pH (Figs. 4g, 6). Concentrations and loads of dissolved Ca and Mn increased within the wetlands because of continuous groundwater seepage into the wetlands and, to a lesser extent, 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/year, and the increase in Ca load was 23.8 t/year 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 estimated cost ($1,584/t) compared to the other treatment systems (Table 3). If the cost for lime was considered, the estimated treatment cost 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°C exhibited by the Rowe Tunnel discharge and the Lorberry Creek diversion well effluent (Supplementary material), the temperature of the wetland effluent ranged widely (Fig. 6 and Supplementary material). During summer, the temperature of the wetland effluent increased to 25.4°C, which greatly exceeds the upper limit of 18.7°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 et al. 2010), the potential for adverse temperature effects would need to be considered in possible plans for additional wetlands to treat other AMD sources, such as the Shadle or Pantherhead discharges (Fig. 1).

Limestone-Compost-Based Wetlands on Lower Rausch Creek (WLR; E3-1–E3-2)

The wetlands constructed in December 1997 on Lower Rausch Creek impounded stream flow within a highway fill area that was underlain by boulders and was prone to losing water. Hence, because of seepage losses and evaporative losses, the flow rate exiting the wetlands typically was less than that entering the wetlands (Fig. 6 and Supplementary material). If the decreased flow resulted 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 concentrations of net acidity of −24.8 to 2.3 mg/L $CaCO_3$, pH 6.2–7.8, and widely variable concentrations of dissolved SO_4 of 62–207 mg/L, and Fe, Al, and Mn, ranging from <0.05 to >1.5 mg/L (Supplementary material). In comparison, the water downstream from the wetlands exhibited significantly lower concentrations and loads of net acidity, dissolved SO_4, and dissolved and total Fe, Al, and Mn, plus lower pH (Figs. 4h, 5h, 6). Although the concentrations of dissolved and total Ca were equivalent for the influent and effluent, the load of Ca decreased through the wetlands (Supplementary material).

The Rausch Creek wetlands removed a median acidity load of 3.1 t/year (Table 3). In contrast with the other treatment systems that exported Ca, the Rausch Creek wetlands removed 27.0 t/year 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 estimated cost ($7,783/t) (Table 3). 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 (Fig. 6 and Supplementary material). 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 substrate. 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 occurring 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 of samples collected during 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).

Discussion

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 that the ALD on the Buck Mountain discharge (treatment ADB at site A1 in Fig. 1) near the headwaters of Swatara Creek had the greatest overall benefit. This ALD, which had been in service for more than 10 years, consistently exported an annual load of $CaCO_3$ greater than 26 t/year, equivalent to the acid removed, and produced significant improvement in pH of downstream water for relatively low estimated cost. Compared to the 29 ALDs evaluated by Ziemkiewicz et al. (2003), the median flow rate treated by the Buck Mountain ALD was two times greater than the highest they reported, and the acid-removal efficiency was near the median value for other ALDs. However, the estimated cost for the Buck Mountain ALD was greater than 75% of the ALDs evaluated by Ziemkiewicz et al. (2003), reflecting added expenses incurred for enlarging the Buck Mountain treatment system in 2001 and 2005.

The other treatment systems in the upper Swatara Creek Basin had treatment efficiencies and estimated 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 estimated 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 rapidly changing, moderate to high flows. Because storm flow 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 stream flow 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 storm flow 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}, $Mn^{III–IV}$, and Al oxyhydroxides at

near-neutral pH, slow rates of oxidation limit passive treatment and metal-removal efficiency (e.g. Cravotta 2007; Watzlaf et al. 2004). 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 this 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 estimated 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. 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 ALD 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.

Summary and Conclusions

A variety of treatment systems 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; the eight systems evaluated in this paper were installed from 1995 through 2001. Periodic measurements of flow rate and chemical concentrations upstream and downstream of each system indicated that all eight were effective at decreasing the acidity load. However, each system had unique influent and effluent characteristics, and the treatment performance varied considering the acid load removed relative to the size of the treatment system and the estimated 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 ALD was effective at removing acidity at relatively low cost; (5) the two sets of limestone diversion wells were relatively expensive but effective for treating stream water 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 storm flow conditions. Generally, storm flow was acidic, and, as stream flow 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 storm flow 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 would need to 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 stream water 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, could be improved and considered by resource managers and other stakeholders involved in mine drainage remediation.

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