CHALLENGES IN DESIGN AND CONSTRUCTION OF A LARGE MULTI-CELL PASSIVE TREATMENT SYSTEM FOR FERRUGINOUS LEAD-ZINC MINE WATERS[1]

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Abstract: Artesian discharges of net alkaline, ferruginous waters from abandoned underground lead-zinc mines cause considerable surface water degradation at the Tar Creek Superfund Site, part of the historic Tri-State Mining District of Oklahoma, Kansas and Missouri. Two perennial borehole discharges, identified as the lowest elevation mine water discharge points in the district, have flowed unabated for almost 30 years and considerably degraded the physical, chemical and biological integrity of a first-order tributary to Tar Creek. Based on a comprehensive water quality and quantity characterization study, a large multi-cell passive treatment system was designed to receive approximately 1000 L/minute of mine water flowing from these abandoned boreholes (pH 5.95 ± 0.06, total alkalinity 393 ± 18 mg/L $CaCO_3$, total acidity 364 ± 19 mg/L $CaCO_3$, Fe 192 ± 10 mg/L, Zn 11 ± 0.7 mg/L, Cd 17 ± 4 ug/L, Pb 60 ± 13 ug/L and As 64 ± 6 ug/L). The objectives of this project include: i) remediation of polluted mine waters to acceptable quality for maintenance of the receiving water body aquatic community, ii) demonstration of the first mine water treatment facility of any kind in the Tri-State Mining District, and iii) technology transfer to speed application of this technology to other locations. The passive treatment system includes an initial oxidation pond followed by parallel treatment trains (to facilitate research and experimentation) consisting of aerobic wetlands, vertical-flow bioreactors, re-aeration ponds (with active aeration via wind and solar power) and horizontal-flow limestone beds. Waters from the parallel trains are recombined in a polishing wetland prior to final discharge. Total design surface water elevation change in the entire system is approximately 1.8 m. Prior to system implementation, the abandoned boreholes required rotosonic over-drilling to establish hydraulic control. In addition, diversion of storm water flows from an approximately 470-ha upgradient watershed was necessary. During construction, a third mine water discharge was discovered and incorporated into the design. This system represents a state of the art ecological engineering research site for passive treatment of mine waters.

Additional key words: metal mining, acid mine drainage, natural treatment systems

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Introduction

Abandoned hard-rock mine sites pose considerable ecological and human health risks (Environmental Protection Agency 2004a; Limerick et al. 2005). Although a single accepted definition of what constitutes an abandoned hard-rock mine site is lacking, the Government Accountability Office (2008) estimated that 161,000 sites existed in 13 western states (AK, AZ, CA, CO, ID, MT, NV, NM, OR, SD, UT, WA, and WY). These sites included at least 332,000 features that "may pose physical safety hazards" and 33,000 locations that have "degraded the environment" in some way. Using a different definition, EPA (2004a) estimated that over 550,000 abandoned hard-rock mine sites containing more than 50 billion tons of untreated mining wastes exist across the U.S. In addition, 63 abandoned hard-rock mine sites are on the National Priorities List (NPL) and at least another 93 are under review for listing (EPA 2004b). Clean up costs were estimated to be $7-$24 billion for just 156 abandoned hard-rock mine sites (approximately 12 times the total annual U.S. EPA Superfund budget). In 1998, the Western Governor's Association published a review of abandoned hard-rock mine problems that identified the need to address the complex environmental policy issues associated with these problems.

Among the many environmental problems at hard-rock mining locations, degraded surface and ground water quality is a major concern. Contaminated mine drainage is characterized by elevated concentrations of metals (Fe, Zn, Cd, Cu, Ni, Mn, Al, Pb, etc.), acidity and sulfate (e.g., Younger et al. 2002). Relatively laborious and cost-intensive active treatment technologies are not viable options for most abandoned mines (e.g., Younger et al. 2002; Watzlaf et al. 2004). Passive treatment technologies, i.e., those that rely on natural biogeochemical, physical and microbiological processes to ameliorate mine drainage problems, may provide viable treatment alternatives to traditional treatment technologies. Passive treatment systems are an application of the principles of ecological engineering, defined as "the design of sustainable ecosystems that integrate human society with its natural environment for the benefit of both" (Mitsch and Jorgensen 2004). As such, the overall goals of ecological engineering projects are the restoration of disturbed ecosystems and development of new sustainable ecosystems having both human and ecological value. Passive treatment of abandoned mine drainage therefore fulfills both the ecosystem restoration and development goals of this emerging field.

This paper describes the implementation of an innovative passive treatment system design for abandoned ferruginous Pb-Zn mine waters at the Tar Creek Superfund Site of Oklahoma. The multi-cell system is the first application of any full-scale mine water treatment at this Superfund site.

Study Site

The study site is located in the Tar Creek Superfund Site, part of the Picher Mining Field and the northeastern Oklahoma portion of the former Pb and Zn mining area known as the Tri-State Mining District (Fig. 1). In this area, the ore deposit consists of Pb and Zn sulfides associated with cherty carbonate host rock (McKnight and Fischer 1970). The principal ore host stratum is the Boone Formation, composed of fossiliferous dolomite, limestone and nodular chert (Luza 1983). Principal ore minerals are sphalerite and galena, with secondary concentrations of chalcopyrite, enargite, luzonite, marcasite, pyrite, and barite (Playton et al. 1980). Significant quantities of Pb and Zn were produced from the Tri-State District from the 1890s through the 1960s. By the late 1950s, depressed global markets resulted in the suspension of most mining operations. By the early 1970s when mining ceased, almost 2 million tons of Pb and 9 million tons of Zn had been produced (McKnight and Fischer 1970). The Tar Creek site was proposed for the Comprehensive Environmental Response, Compensation and Liability Act (Superfund) NPL in 1981 and received final listing in 1983. Nearby are the Cherokee County Superfund Site (Kansas) and the Oronogo-Duenweg Mining Belt and Newton County Mines Superfund Sites (Missouri).

In the Oklahoma portion of the district, approximately 1,000 ha are underlain by underground mines in all or part of 47 sections (Luza 1983). During mining, large capacity dewatering operations pumped approximately 50,000 m^3 of water per day from the mines (Reed et al. 1955). Upon decline and cessation of mining, groundwater began to accumulate in the mine voids. Approximately 94 million cubic meters of contaminated water exist in underground voids. By late 1979, metal-rich waters began to discharge into Tar Creek and its tributaries. The first documented discharge of mine drainage was at a location near southeast Commerce, OK denoted by the Oklahoma Water Resources Board (1983) as Site 14 (subsequently identified for passive treatment implementation and the subject of this paper).
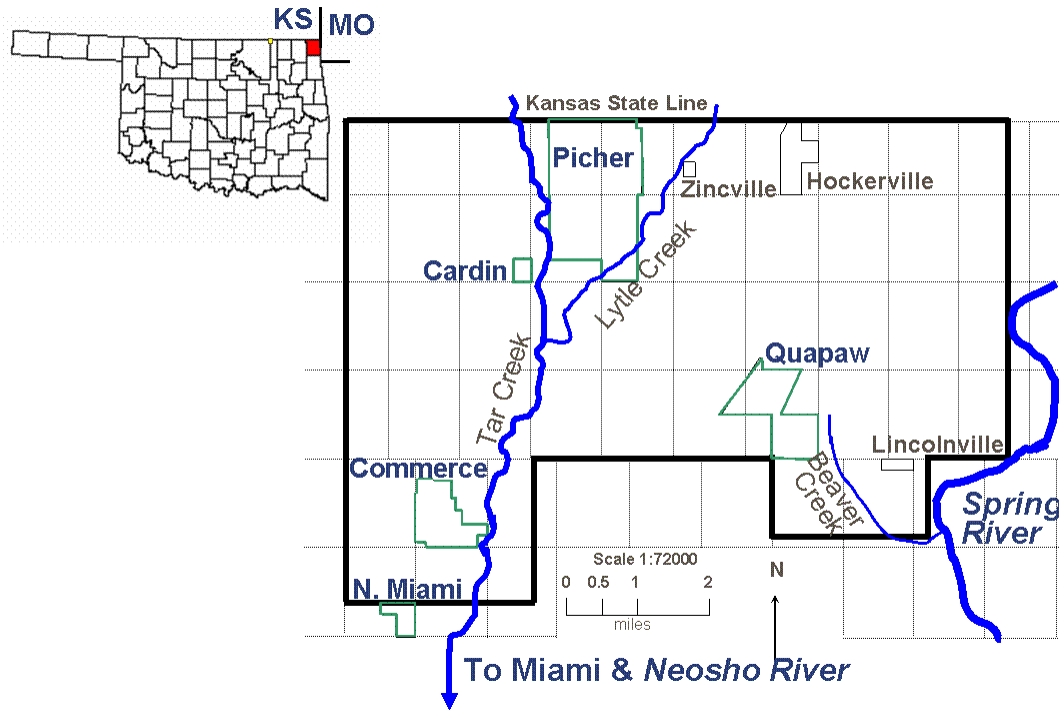
Figure 1. Schematic diagram of the Tar Creek Superfund Site, Oklahoma; inset state map of Oklahoma shows location of Ottawa County.

In general, water quality of mine drainage discharges in the Tar Creek watershed was originally characterized in the mid-1980s as follows: pH 3.6-5.7, 80 ug Pb/L, 154,000 ug Zn/L, 80 ug Cd/L and 331,000 ug Fe/L (EPA, 1999). More recently, several authors (Nairn et al. 2001, 2002, 2007, 2008; Nairn 2002; Coffey and Nairn 2003; DeHay et al. 2003; Iverson and Nairn 2003) documented lesser contaminant concentrations in mine drainage discharges and stream waters than those occurring soon after the discharges began. However, Fe, Zn, Pb and Cd concentrations were still greater than National Recommended Water Quality Criteria (NRWQC) for maintenance of aquatic communities (EPA 2006). It is reasonable to expect these values will continue to be above the NRWQC for decades if not treated.

The passive treatment system described in this paper addresses mine drainage discharges near Commerce (historic site 14; Fig. 2; Table 1). Periodic data collection efforts at this site began in 1998, with regular monthly sampling beginning in 2004 and continuing to the present. This area of the Tar Creek Superfund Site has a limited amount of waste rock and tailings impacts but substantial artesian mine drainage discharges. The targeted discharges (SA and SB)

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have circumneutral pH (5.96±0.06) with total alkalinity of 405±13 mg/L as $CaCO_3$. Combined flow rates of these discharges are 400-700 L/minute.

Figure 2. Aerial photograph of study site prior to passive treatment system construction showing location of mine drainage discharges SA and SB, receiving stream and proximity of local residential and commercial developments.

Conceptual Design

Given the nature of the target discharges (net alkaline with elevated Fe, Zn, Pb and Cd concentrations), a multi-process unit conceptual design was developed. Individual process unit designs focused on specific water quality improvement goals, e.g., Fe oxidation, solids settling, metal sulfide formation, etc. In addition, an identical parallel treatment train approach was deemed appropriate for at least two reasons. First, the parallel trains allow for simultaneous performance of necessary maintenance and continued treatment (by taking one set of process units off-line while allowing continued function of the other set). Second, given the research focus of this site, the parallel trains allow experimental manipulations to be conducted. For example, the identical trains may be exposed to different mass loadings. The conceptual design process identified six distinct process units (Table 2).

Table 1. Summary water quality data for mine drainage discharges SA and SB. All data are mg/L (n = 78-90)

	Discharge SA		Discharge SB	
	mean	standard error	mean	standard error
Al	0.63	0.2	0.38	0.08
As	0.077	0.008	0.068	0.001
Ca	749	11.13	742	9.42
Cd	0.017	0.0004	0.017	0.0002
Co	0.093	0.002	0.092	0.001
Cr	0.003	0.0005	0.002	0.0001
Cu	0.007	0.001	0.005	0.001
Fe	203	4.25	198	3.22
Mg	203	3.6	202	3.14
Mn	1.69	0.029	1.68	0.024
Na	100	2.12	97.07	1.72
Ni	0.966	0.016	0.977	0.012
Pb	0.045	0.004	0.057	0.004
Zn	11.35	0.232	10.71	0.203
SO_4^{-2}	2154	53	2165	62

Table 2. Summary of final conceptual design process units, primary targeted water quality parameters and design function.

Process unit	Targeted parameter	Function
Oxidation pond	Fe	• Oxidation, hydrolysis and settling of iron oxyhydroxide solids • Trace metal sorption
Surface-flow wetlands/ponds	Fe	• Solids settling
Vertical-flow bioreactors	Zn, Pb, and Cd	• Retention of trace metal sulfides via reducing mechanisms
Re-aeration ponds	Oxygen demand and odor	• Wind- and solar-powered re-aeration • Stripping oxygen demand and H_2S • Adding O_2
Horizontal-flow limestone beds	Zn, Mn and hardness	• Final polishing of Zn as $ZnCO_3$ • Final polishing of Mn as MnO_2 • Adding hardness to offset bioavailability of any remaining trace metals
Polishing pond/wetland	Residual solids	• Solids settling • Photosynthetic oxygenation • Ecological buffering

Process Unit 1: Oxidation Pond

Aerobic Fe removal is a well-documented process for net alkaline ferruginous mine waters (e.g., Hedin et al 1994; Younger et al 2002, Watzlaf et al 2004; Hedin 2008). An initial oxidation pond was designed to receive untreated artesian mine waters, oxidize and hydrolyze elevated Fe^{2+} concentrations and settle iron oxyhydroxide solids. The conceptual design called for a relatively deep (>1 m) cell for solids retention with an appropriate water surface area (sized for approximately 20 g m^{-2} d^{-1} removal) for passive oxygen transfer.

On-site "bucket experiment" field trials (Nairn et al., unpublished data) documented effective oxidative Fe removal for these waters. Six *in situ* treatments were established in triplicate in 40-L containers and analyzed over 24 hours: i) closed, ii) closed with Fe, iii) open, iv) open with Fe, v) open and aerated, vi) open and aerated with Fe. "Closed" treatments were containers covered with a low permeability lid. "Open" treatments were left open (no lid) to the atmosphere. "Aerated" treatments received one small aquarium air stone run continuously throughout the experiment. "With Fe" treatments received a small quantity of precipitated solid iron oxyhydroxide precipitate to examine differences between homogenous and heterogeneous iron oxidation rates.

For purposes of this discussion, mean data for triplicate treatments are plotted (Fig. 3 and 4). Iron concentrations decreased rapidly in aerated treatments from >200 mg/L to <10 mg/L over 24 hours. Open treatments showed some decrease (to ~ 170 mg/L) whereas closed treatments demonstrated little change (to ~ 190 mg/L). Simple aeration resulted in substantially increased Fe removal rates (Fig. 3a).

As expected, dissolved oxygen concentrations increased greatly in the aerated treatments, slightly in the open treatments but did not change in the closed treatments (Fig. 3b). Despite the production of proton acidity by Fe hydrolysis in the aerated treatments, pH increased to almost 8 (Fig. 4a). In open treatments, pH increased slightly and in closed treatments it did not change. The increased pH with increased Fe oxidation and retention indicates the major importance of CO_2 degassing by aeration. Any pH decrease due to the considerable production of proton acidity was outcompeted by the pH increase due to CO_2 loss in these net alkaline waters. Not only did simple aeration add oxygen to the water column, it helped to exsolve CO_2 leading to enhanced treatment performance.
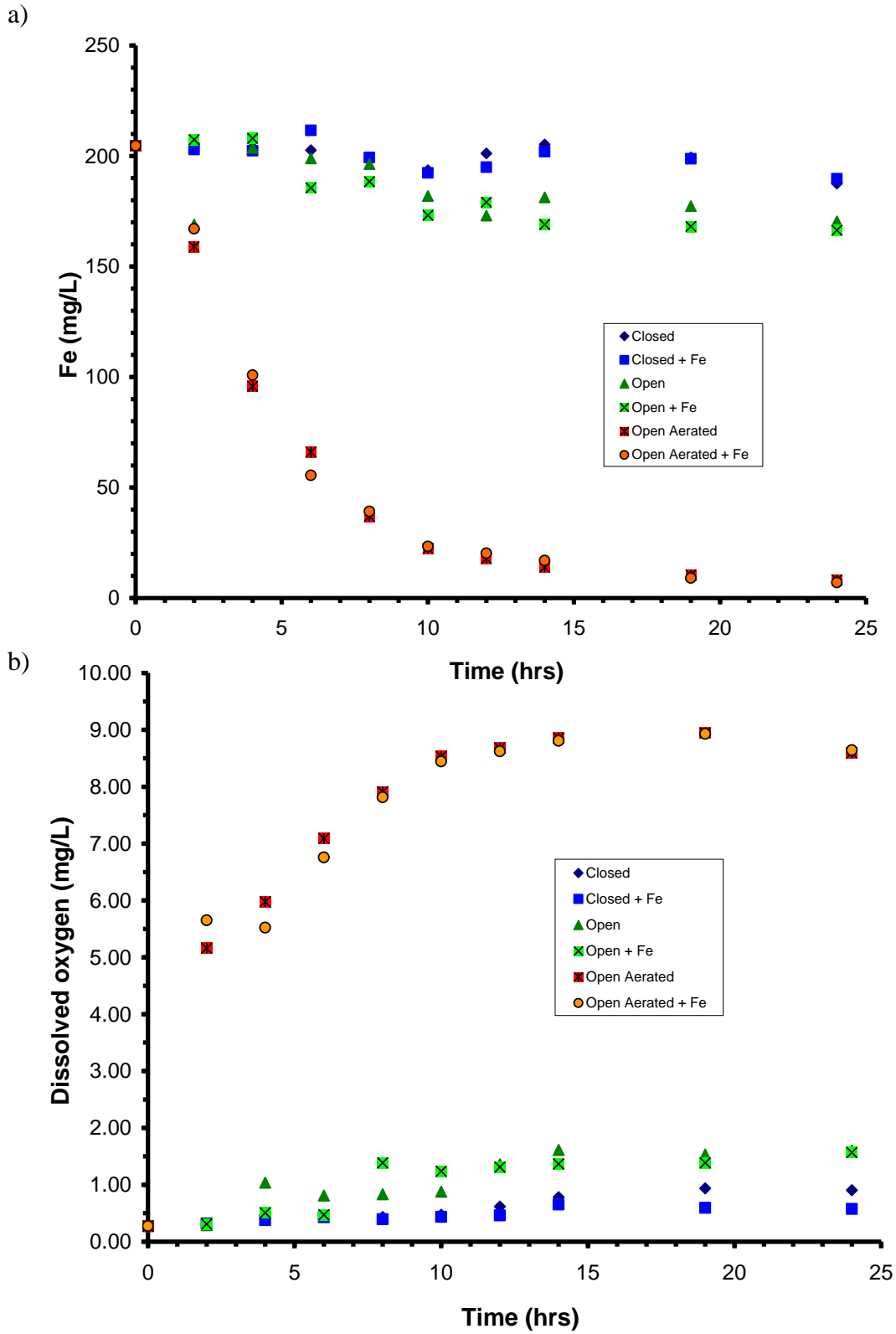
a)

b)

Figure 3. a) Total Fe concentration and b) dissolved oxygen concentration changes in on-site Fe oxidation field trials.
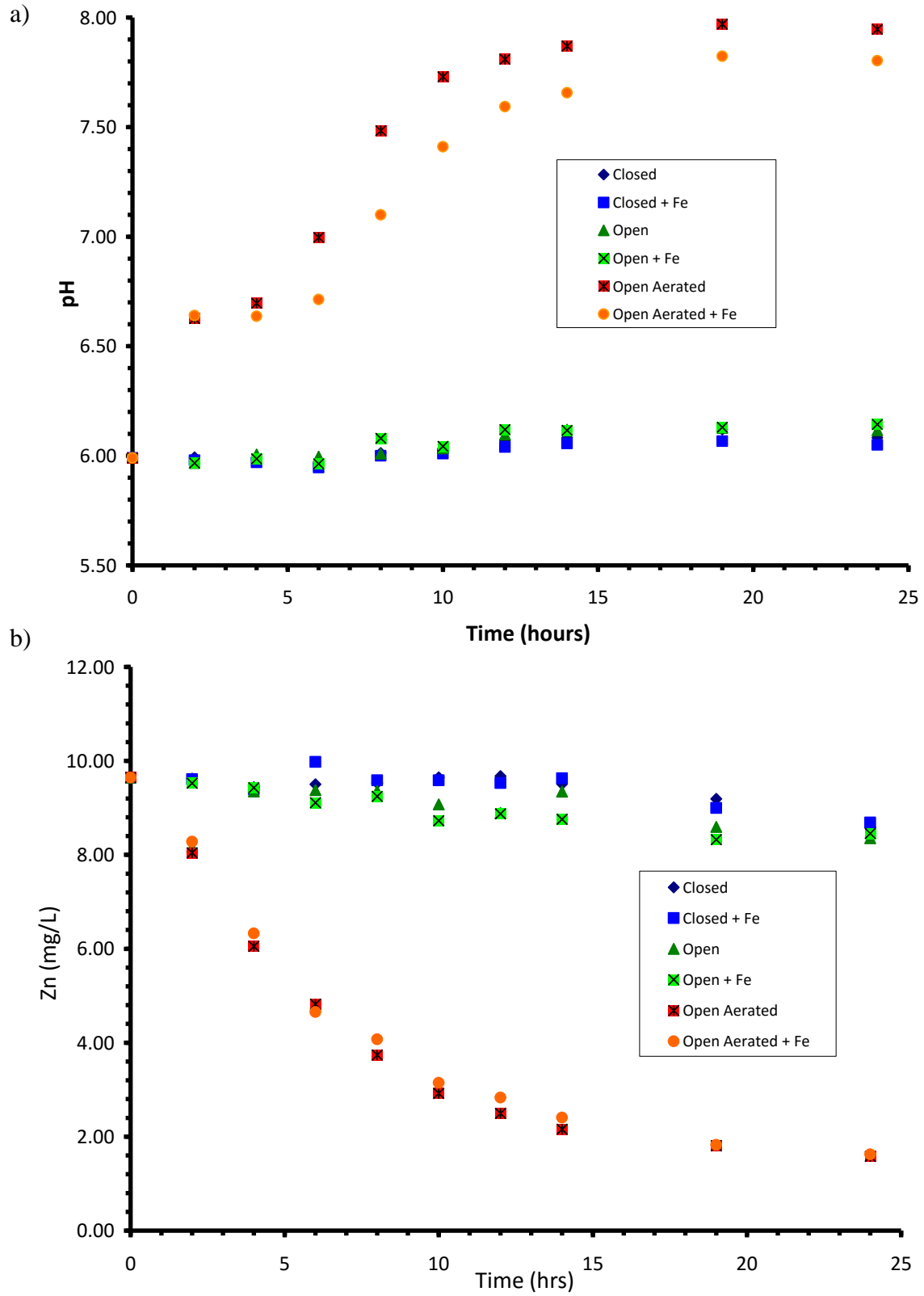
Figure 4. a) pH and b) total Zn concentration changes in on-site Fe oxidation field trials.

In addition, Zn concentrations decreased dramatically in those treatments which exhibited the greatest Fe removal (Fig. 3a and 4b). The affinity of Zn for solid Fe oxyhydroxides, especially at circumneutral pH, is well-documented (Edwards and Benjamin 1989; Dzombak and Morel 1990; Benjamin and Sletten 1996; Carroll et al. 1998). Zinc was likely removed from solution via co-precipitation and sorption. Arsenic, Cd, Ni and Pb exhibited similar trends (data not shown).

Process Unit 2: Surface-Flow Aerobic Wetlands/Ponds

After oxidation and hydrolysis, settling of Fe solids is necessary for effective treatment (Younger et al. 2002; Hedin 2008). Therefore, process unit 2 was designed as a surface-flow aerobic wetland to facilitate solids settling. Emergent vegetation in this unit provides for a tortuous flow path and increased retention time, allowing already oxidized and hydrolyzed Fe solids to settle from solution. The conceptual design for this unit was a relatively shallow (<0.3 m) wetland area for effective plant establishment and growth, along with deeper water pools to facilitate solids settling and retention.

Process Unit 3: Vertical-Flow Bioreactors

Process units 1 and 2 were designed primarily to address elevated Fe concentrations from the mine waters. Although some trace metal removal via sorption and co-precipitation was anticipated, process unit 3 was designed specifically to remove Cd, PB and Zn. Reductive mechanisms are well-established for passive treatment of trace metals (e.g., Jong and Perry 2006; Neculita et al. 2007). Conceptual designs of the vertical-flow bioreactors, which provide reducing environments, included a waste organic substrate layer (e.g., spent mushroom compost) overlying a limestone layer and perforated pipes to facilitate flow through the reactive media.

Laboratory-scale, vertical flow bioreactor column experiments documented trace metal removal for these mine waters (Viswanathan 2005). After initial oxidation in separate units, mine water entered two sets of 15.24-cm diameter Plexiglas columns for a period of almost six months. Bacterial sulfate reduction was documented through both changes in sulfate concentrations (column A: 1493 ± 98 to 749 ± 32 mg/L; column B: 1482 ± 95 to 910 ± 20 mg/L) and presence of sulfate reducing bacteria (10^4 to 10^6 CFU/g) in the organic substrate layer.

Effective Pb and Cd removal occurred in both columns shortly after experimental start-up (Fig. 5) and continued for the duration of the experiment. In addition, autopsy of the columns indicated substantial Cd, Pb and Zn retention (Table 3).
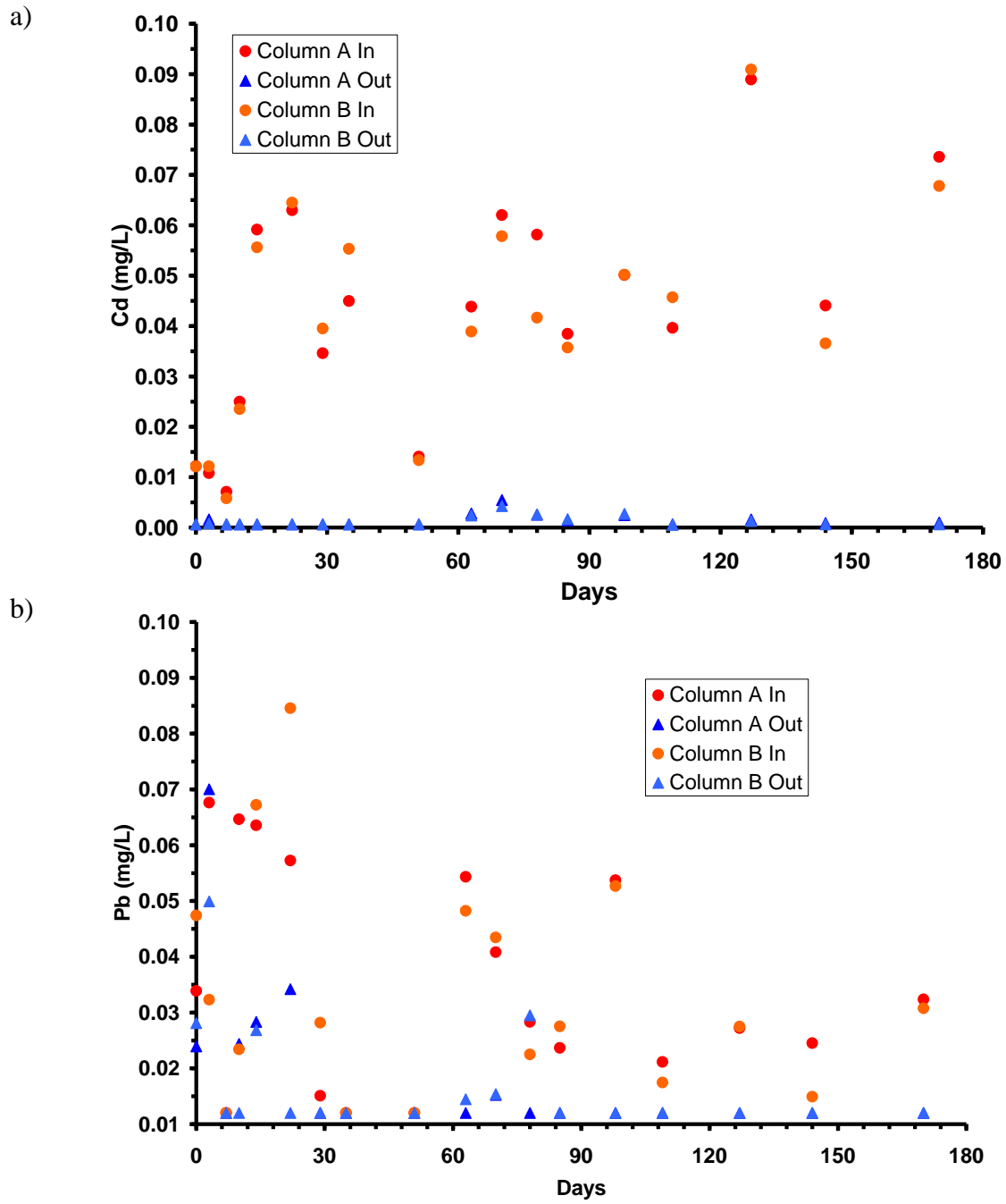
a)

b)

Figure 5. a) Cd and b) Pb concentration changes in vertical-flow bioreactor columns.

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Table 3. Column substrate metal concentrations (mg/kg) for Cd, Pb and Zn (mean ± standard deviation, n=20 for column A, n=15 for column B) after six months of treatment.

	Cd	Pb	Zn
Column A			
mean	2.75	21.04	1058
sd	2.45	4.14	1021
Column B			
mean	2.34	19.78	994
sd	0.70	2.28	503

Process Unit 4: Re-aeration Ponds

After flow though vertical-flow bioreactors, waters are reduced and contain considerable oxygen demand (Porter and Nairn 2008). Therefore, oxygenation of the waters is necessary. Given the proximity of the system to residential and commercial sites, re-aeration units were included in these ponds not to enhance metals treatment but to address the potential for odor problems (mainly H_2S and related sulfidic compounds) and oxygen demand. The conceptual design for this unit was a deep pond with submerged aeration systems powered by renewable energy sources.

Submerged aeration systems are commonly used in aquaculture, lake and reservoir management and lagoon operations to promote mixing and to increase dissolved oxygen levels (e.g., Westerman and Zhang 1997; Boyd 1998; DeMoyer et al. 2003). A typical submerged aeration system consists of a renewable power source that builds pressure in an air compressor. The pressure is released through a valve and line to a submerged diffuser at the bottom of the pond, which produces a "bubble plume" that rises to the surface. The bubbles cause vertical and lateral circulation by entraining water as they rise. The bubble interfaces help to transfer oxygen to the water as they rise as well. At the pond surface, further oxygen mass transfer occurs due to turbulence at the air-water interface. Two power options were identified for renewable-energy based water column re-aeration devices: wind and solar power.

Windmill-powered re-aeration is a well-established technology for aquaculture. According to Superior Windmill Aeration Systems (2009), these systems have the ability to develop a steady stream of air at more than 30 pounds of pressure "enabling deep placement of the diffuser for maximum cleansing action" The towers are typically 12-20 feet in height with the

compressor mounted directly behind the fan blades. A single diaphragm compressor can produce 1.5 cubic feet per minute (cfm) of air and a dual diaphragm compressor can produce 3 cfm, both at a 9 mile per hour (mph) wind speed.

Solar powered units use small solar panels to generate energy for a battery-driven compressor. Because the panels continuously charge the battery, these units will continue to work at night and on cloudy days. According to Solar Aerators (2009), these units can run continuously for two weeks of cloudy days. Using a single 0.5 amp, 18-volt solar panel and a standard 12-volt automobile battery, these units can pump 1.0 cfm; use of multiple diffusers can pump up to 4 cfm.

Process Unit 5: Horizontal Flow Limestone Beds

To further ensure effective water quality improvement, horizontal flow limestone beds (HFLBs) were included in the design to perform three functions: final Zn polishing, final Mn polishing and increased water hardness to offset bioavailability of any remaining trace metal concentrations. The conceptual design included relatively deep limestone beds (>1 m) with water levels to be controlled below the surface of the stone, providing for a minimum 14-hour retention time.

Limestone-based reactors have been documented to effectively retain Zn via various mechanisms given the proper pE-pH conditions (Cravotta and Trahan 1999; Nuttall and Younger 2000). Elevated partial pressures of CO_2 in this system promote carbonate dissolution and production of greater than atmospheric equilibrium concentrations of alkalinity in the substrate. Although Mn is of limited concern in these waters, similar results for Mn retention have been documented (e.g., Robbins et al. 1999; Denholm et al. 2008).

For trace metals, aquatic life criteria based on EPA NRWQC (EPA 2006) include chronic (Criterion Continuous Concentration or CCC) and acute (Criterion Maximum Concentration or CMC) values for in-stream metal concentrations that are both contaminant and hardness-specific. In general, as hardness increases, the toxicity of a particular metal at a given concentration decreases. Untreated waters discharging from the site exceeded these criteria for Fe, Zn, Pb and Cd concentrations. Therefore, significant decreases in metal concentrations at the point of discharge coupled with increases in hardness in the HFLBs are necessary to maintain the receiving water body aquatic community, a first order tributary to Tar Creek.

Process Unit 6: Polishing Pond/Wetland

The polishing pond/wetland was designed as the final process unit. Waters are designed to be retained for residual solids settling via vegetative filtration and re-aeration via passive photosynthetic means. This process unit was also designed to provide habitat function and to serve as a buffer between the passive treatment system and receiving water body and its associated riparian zone, serving as an ecotone to blend the treatment system into the surrounding native environment.

Engineering design

Through a competitive bidding process, a design/build engineering contract was awarded to CH2M-Hill using funds provided by the U.S. Environmental Protection Agency and U.S. Geological Survey. The intent of this project was to develop engineering plans and specifications for the design and construction of the passive treatment system, build the system and provide as-built construction documents. Engineering design, based on the conceptual designs and taking into account on-site conditions, was undertaken as an iterative process between the design firm, the University of Oklahoma and EPA. To accommodate site constraints, multiple adjustments were made to the system layout. Most notable of these included the configuration of the system as two parallel flow paths to allow estimation of variance in performance and to perform maintenance, consolidation of the layout into a more compact "footprint" area to comply with construction requirements adjacent to an existing utility corridor and to minimize impacts to adjacent landowners, thereby creating a flatter hydraulic profile. Rotosonic over-drilling of both mine drainage seeps was completed 18 months prior to passive treatment system construction, thus providing hydraulic control and allowing additional investigations to further quantify flow rates and variability.

Design and construction tasks included capture and control of the two known artesian mine drainage discharges, diversion of storm water flows from a 470-ha up-gradient watershed, implementation of all passive treatment process units including water conveyance structures, and provision of as-built documents. Design performance was estimated using inflow concentrations of 192 mg/L Fe, 11 mg/L Zn, 17 ug/L Cd and 60 ug/L Pb, and Fe removal rate of 20 g m^{-2} d^{-1} with target Fe effluent of 1 mg/L and assuming net alkaline conditions, a discharge rate of 1000 L/minute, available site area of approximately 3.6 ha and treatment area of 2 ha.

Construction began in July 2008 and was completed in late November 2008. Issues requiring

resolution during construction and startup included the incorporation of a third seep (Seep D) into the inflow oxidation pond, removal of debris and existing iron oxide muck, delays throughout the course of construction caused by a record rainfall during the summer and related site water management issues, and changes in organic substrate requirements. The third seep was isolated as a distinct inflow into the first process unit and did not result in inflow discharge rates or mass loadings substantially different from initial designs. Based on additional laboratory work, the vertical-flow bioreactors received a mixture of 45% spent mushroom compost, 45% hardwood chips and 10% manufactured limestone sand in the organic substrate layer. The total cost of design and construction totaled $1,196,000.

The completed system (Fig. 6) includes 10 distinct process units with a single initial oxidation pond (cell 1) followed by parallel surface-flow aerobic wetlands/ponds (cells 2N and 2S), vertical-flow bioreactors (cells 3N and 3S), re-aeration ponds (Cells 4N and 4S) and horizontal flow limestone beds (Cells 5N and 5S), and a single polishing pond/wetland (Cell 6). Mine water was diverted into the passive treatment system for the first time on December 2, 2008.

Performance Monitoring

Given recent system start-up, limited performance data are available at this time. Monitoring began in January 2009. During normal operation, the function of the passive treatment system will be evaluated from several perspectives. The overall performance of the passive treatment system will be evaluated by regular collection and analysis of water quality and quantity samples at the inflow and outflow of each cell. Input/output analyses, although essential, provide only a "snap-shot" of passive treatment system performance. Therefore, detailed evaluations of water quality, substrates and pore waters are necessary to "open the black box" of passive treatment. Spatial water quality and underlying substrate/pore water collections will be conducted to evaluate the dominant biogeochemical processes contributing to contaminant retention in each of the cells. Substrate cores will be obtained and pore waters extracted for analysis. Additional cores will be incrementally sub-sampled to allow the development of detailed substrate profiles, including sequential extraction, total metals digestion, Toxicity Characteristic Leaching Procedure (TCLP; EPA Method 1311) and x-ray diffraction analyses. Dominant microbiological activity and species will be determined, taking into account the likelihood of a bacterial assimilation period after first flooding of the system. Specific microbial groups expected within
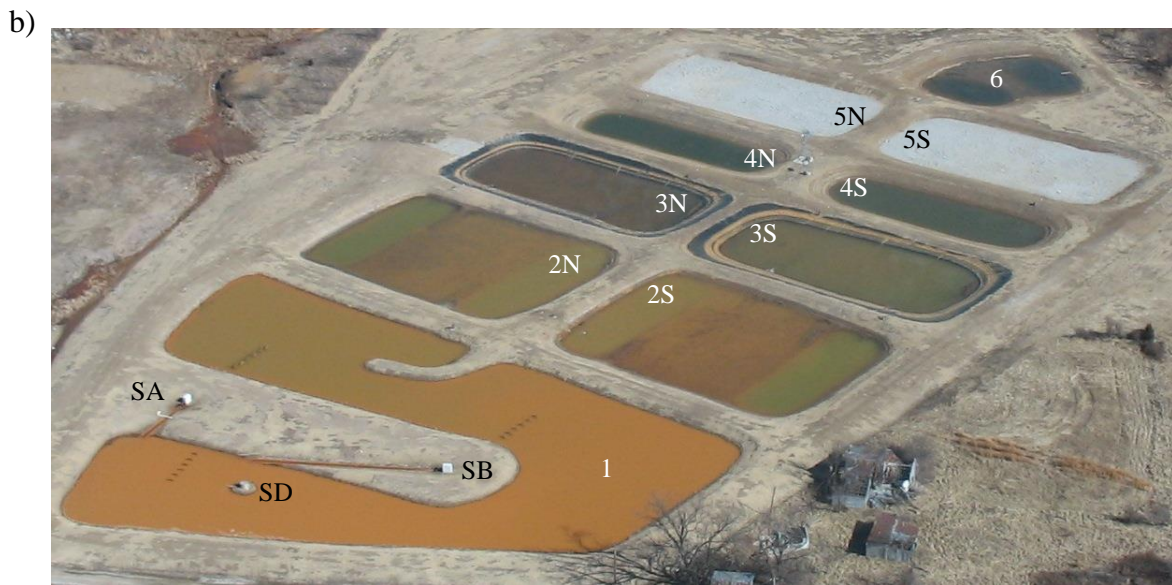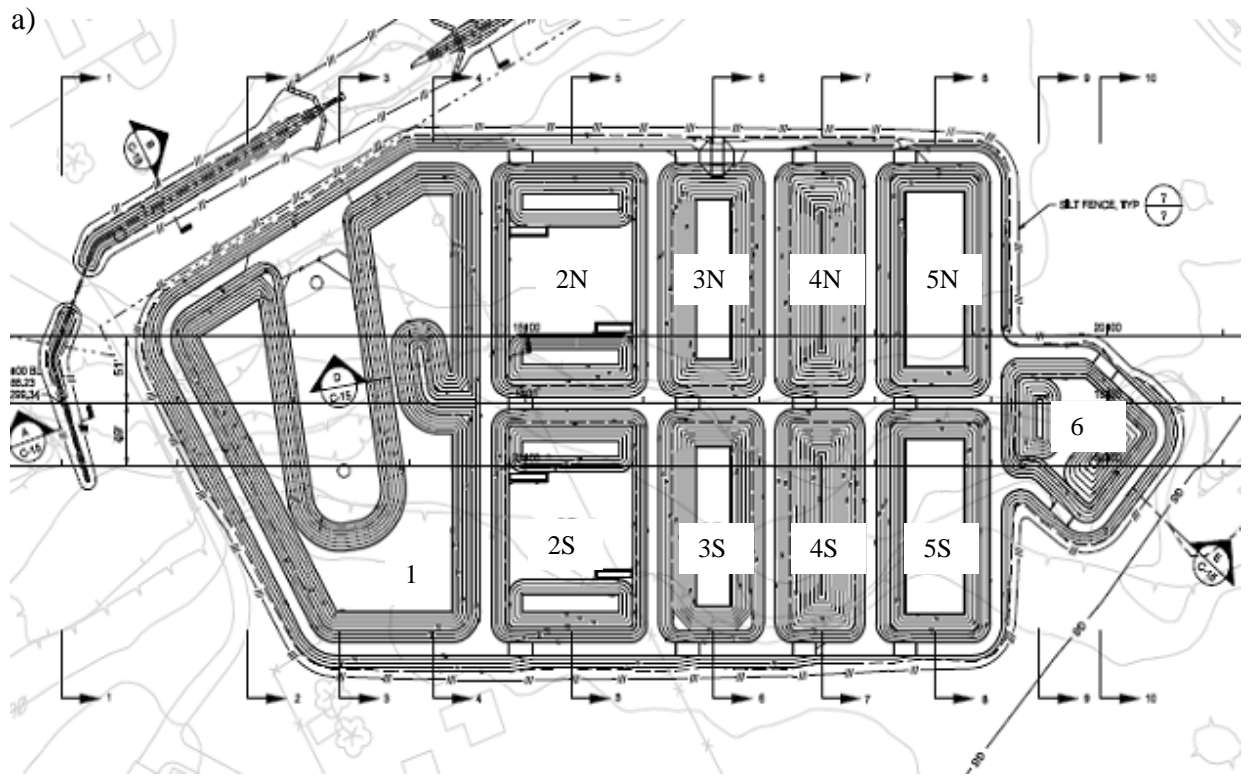
Figure 6. a) Plan view of engineering design and b) aerial photograph of completed system taken February 4, 2009.

the passive treatment system include (but are not limited to) sulfate-reducing bacteria, iron-reducing bacteria, iron-oxidizing bacteria, anaerobic/aerobic fermenters, and methanogens. Evaluation of overall ecosystem structure and function will be conducted using vegetation development as a robust and comparable indicator and will include above- and below-ground sampling.

In conjunction with regular water quality data collection within the passive treatment system, water quality and discharge rates of the receiving waters will be characterized both upstream and downstream of the system effluent, and at a reference stream of comparable size outside of the Tar Creek watershed but within the larger Neosho River basin. In addition, fish and macroinvertebrate communities will be evaluated to establish a baseline of the existing native communities and to monitor changes in the communities over time, as the passive treatment system comes on line. Because the passive treatment system is designed to retain and sequester metal contaminants that currently freely enter the environment in the form of mine drainage flowing from abandoned boreholes, the potential for bioaccumulation will be evaluated. An observational protocol has been established to determine the risk to waterfowl and other birds consuming vegetation and/or benthic macroinvertebrates.

Conclusions

Based on an extensive field data collection effort, laboratory and field experiments and a cooperative effort between academia, the private sector and government, the first full-scale passive treatment system in the Tri-State Mining District was designed and constructed. This system represents a start-of-the-art ecological engineering field research site. Although only recently placed into operation, it is anticipated that this site will provide data contributing to the enhancement of future passive treatment system designs.

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