

Evaluating remediation alternatives for mine drainage, Little Cottonwood Creek, Utah, USA

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Abstract The vast occurrence of mine drainage worldwide, documented in descriptive studies, presents a staggering challenge for remediation. Any tool that can move beyond descriptive study and helps to evaluate options for remediation in a way that maximizes improvements to the water quality of streams and minimizes cost of remediation could save valuable resources and time. A reactive solute transport model, calibrated from two detailed mass-loading studies in Little Cottonwood Creek (LCC), Utah, provides a tool to evaluate remediation options. Metal loading to LCC is dominated by discharge from two mine drainage tunnels. Discharge from an upstream tunnel has been treated by a fen to reduce metal loading. Discharge from the downstream tunnel (WDT) can be controlled because of a bulkhead that creates a mine pool. Simulations of remedial options for three compliance locations suggest that the water-quality standards for Cu and Zn at upstream and downstream compliance locations are met using various combinations of fen treatment and WDT regulation, but the complete compliance at the middle compliance location requires the highest level of fen treatment and the greatest regulation of WDT discharge. Reactive transport modeling is an useful tool for the evaluation of remedial alternatives in complex natural systems, where multiple

hydrologic and geochemical processes determine metal fate.

Keywords Mine drainage · Remediation · Transport modeling · Water-quality standards · TMDL

Introduction

In recent years, some mine drainage problems have been addressed by groups of stakeholders that bring together mine and property owners, environmental groups, and concerned citizens with Federal, State, and local agencies. Concerns about discharge of metal-rich water from historical mine drainage tunnels to Little Cottonwood Creek (LCC), near Salt Lake City, Utah, have been addressed by such a stakeholder group. Under the direction of the Utah Division of Water Quality, a total maximum daily load (TMDL) study was conducted to evaluate loads for dissolved zinc (Zn) and to define the extent of treatment that would be necessary to achieve instream water-quality standards (Shepherd Miller 2002). The study indicated that achieving the standards would be expensive, much beyond the resources of the stakeholder group, and so a study was proposed to use reactive solute transport modeling to test combinations of treatments that could provide the least costly alternative.

Reactive solute transport modeling has been combined with field experiments to evaluate remediation options in other settings. Runkel and Kimball (2002), for example, simulated the potential effects of an active treatment system located in the headwaters of Mineral Creek, Colorado. Simulation results for two hypothetical treatment plans indicated a decrease in dissolved concentration for most metals in response to remediation. Results for dissolved lead, in contrast, indicated the potential for remedial

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actions to adversely affect water quality, as the second remediation plan resulted in a concentration increase. This non-intuitive result occurs because of a decrease in iron colloids, an important sorbent for dissolved lead. Ball et al. (2004) evaluated numerous remedial options for the Summitville mine site located in south-central Colorado. Study results are being used by the State of Colorado to redesign and/or replace an existing treatment system at the mine site. Walton-Day et al. (2007) simulated the decrease in zinc load at the mouth of Cement Creek, Colorado, that would result from hypothetical combinations of remediation on Federal lands in the catchment. Through a process of calibrating simulations of a mine-impacted stream and then eliminating those sources that can be attributed to mining, Runkel and Kimball (2007) have used modeling to estimate pre-mining concentrations of metals for Red Mountain Creek, Colorado. The general approach of combining field experiments with reactive solute transport modeling is discussed by Kimball et al. (2003).

The present work combines field experiments that were designed to evaluate potential remedial options with simulation modeling to predict resulting water quality from various combinations of “treatment.” Results of this study are highly relevant to the TMDL process that is occurring for LCC, but have a wider application because this approach could be implemented in many areas affected by mine drainage to evaluate remediation options that could save money.

Methods

Study area and experimental setting

Little Cottonwood Creek begins at about 3,400 m in the Wasatch Mountains and descends more than 2,000 m in the study reach, resulting in a cascading stream with many pools and riffles. The study reach extends from the LCC headwaters near the town of Alta Utah, to the U.S. Geological Survey (USGS) stream gage near Tanners Flat (gage number 10167450, Table E1 of Electronic Supplement). Major sources of metal loading to LCC were identified as part of a mass-loading study in 1998 (Gerner et al. 2001; Kimball et al. 2001). The principal sources were two mine drainage tunnels that discharge directly into LCC. The Wasatch Drainage Tunnel (WDT, Fig. 1) was the principal source of loading for many metals, particularly for Zn. The WDT combines the drainage from many mines and discharges to the stream in at least three places. The most upstream discharge, when the mine pool behind the bulkhead is deeper than 91 m, is through fractures and a small tunnel, the Superior Point mine (2,470 m; Fig. 1; Douglas Evans, Salt Lake County Water conservancy District,

personal communication 1999). A second inflow at 2,926 m discharges WDT water that is used to cool the electrical generation turbine at Snowbird Ski Resort. Finally, a controllable portion of the WDT discharge enters the stream in two pipes at 3,068 and 3,069 m. The combined Howland–Columbus–Rexall (CCR, Fig. 1) discharge was the greatest source for loading of Cu and the second greatest source for Zn. During the 1998 study, the discharge from the CCR was being treated by a fen system (Jensen and Karimi 2001). Seasonal variability of metal loading was investigated by sampling downstream from the principal metal loading inflows in 2001. These samples included discharge, and both dissolved (10,000 Da-filtered) and colloidal metal concentrations to calculate the seasonal loads of metals.

To quantify changes in the distribution and magnitude of metal loads to LCC, the USGS conducted a mass-loading study in 2004 for comparison to the 1998 study. The study reach in 2004 was mostly the same as in the 1998 study, however, in 2004 two intentional changes were made to affect the metal loading to LCC. First, the CCR discharge was not being treated by the fen, and second, the drainage from the WDT was intentionally increased to test the maximum loading it could contribute. By draining the pool behind the bulkhead below 91 m, the discharge from the Superior Point mine (2,470 m) was eliminated. This 2004 field experiment provided important test conditions for the modeling study by defining two variables that might be considered for less expensive remediation. The first variable is the passive treatment of the fen, and the second is the controllable discharge from the WDT. Combinations of fen treatment and seasonal variation of WDT discharge might meet TMDL requirements.

Tracer injection and synoptic sampling

A mass-loading approach combines various methods. Data collection for the analysis is based on field methods of tracer dilution (Kilpatrick and Cobb 1985) and synoptic sampling (Bencala and McKnight 1987; Kimball et al. 1994). Data analysis is based on methods of calculating loads to obtain detailed longitudinal profiles of mass loading (Kimball et al. 2002, 2004, 2007).

Mass-loading field experiments have frequently been conducted during late-summer base-flow conditions. This is to minimize the effects of diel variation during snowmelt runoff, to maximize the effects of ground-water inflows on the stream, and to facilitate access to stream and inflow sampling sites. The base-flow loading pattern most likely represents those metal loads that contribute to chronic toxicity, which may be particularly important during late-summer, early-fall, and winter months (Besser et al. 1999). The base-flow pattern may prove to be representative of loading patterns throughout the annual cycle of streamflow

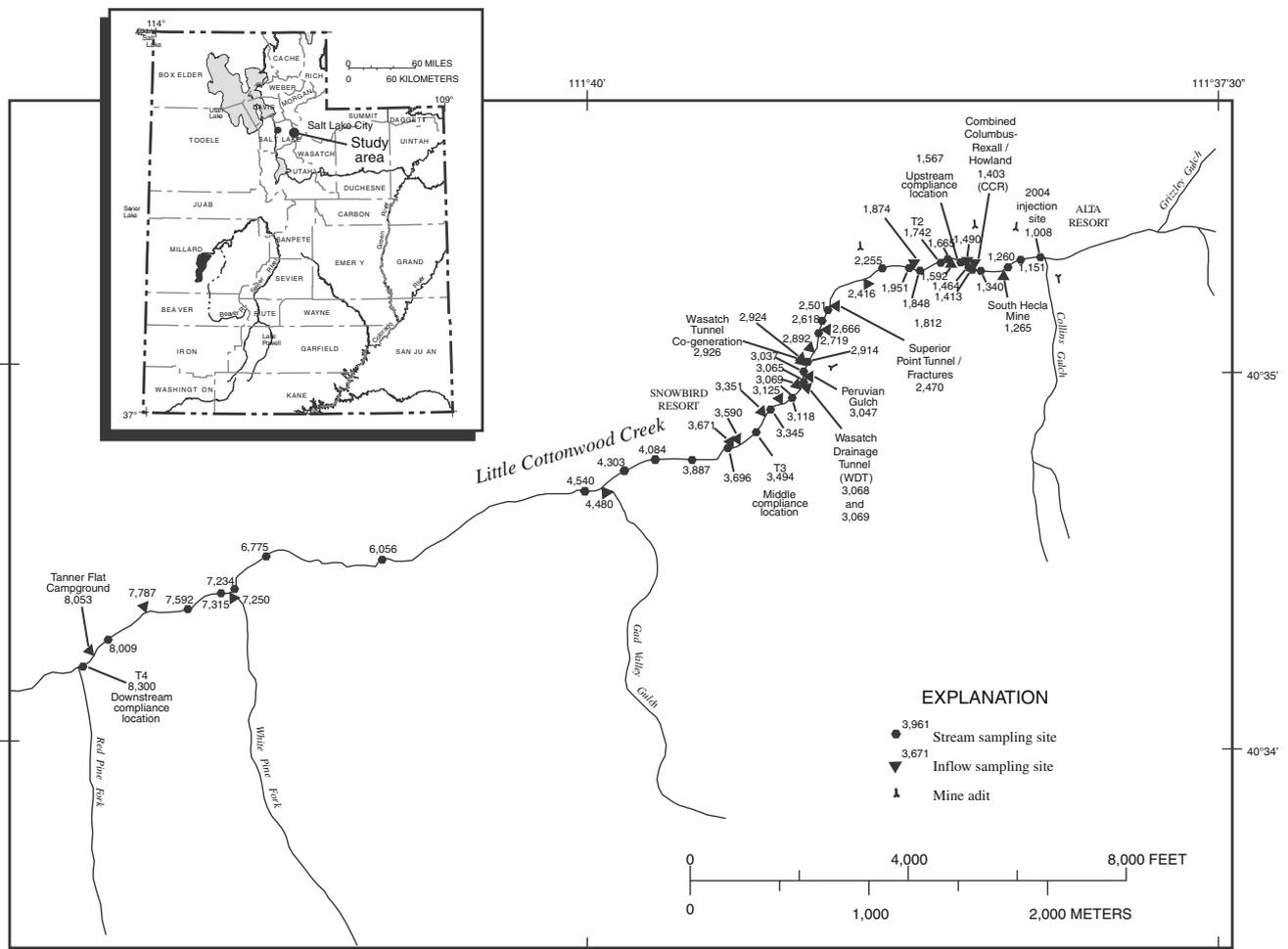


Fig. 1 Location of synoptic sampling sites, Little Cottonwood Creek, Utah

(Leib et al. 2003), but the load calculated for base flow does not represent a mean annual average load to be used for remediation planning.

A careful evaluation of inflows along the study reach was accomplished by walking the entire study reach (Fig. 1), and the inflows found in 2004 were essentially the same as those reported in 1998 (Kimball et al. 2001). Sampling sites for the synoptic study are referenced by the measured distance along the study reach in the downstream direction (Fig. 1). Downstream distances used in 2004 were approximately the same as those used in 1998, with the 2004 tracer injection located at 1,008 m (Table E1 of Electronic Supplement). Inflows are referred to as left and right bank with an orientation looking downstream. Reference to a stream segment means the section of the study reach between two consecutive stream sites, and is referenced by both the upstream and downstream distances, for example the segment 1,375–1,410 m.

Sodium bromide was selected for the injection solution because of the high pH of the stream and the lack of geologic sources of bromide in the watershed. In the

analysis of this experiment, bromide is assumed to be a conservative tracer. The 2004 injection was started September 15 at 0930 hours and continued till September 16 at 1730 hours. The injection rate averaged 2.83 mL/s, and the injectate concentration was 128.5 g/L as bromide. A Campbell CR-10 data logger controlled the pumps by counting pump revolutions and making adjustments to maintain a constant number of revolutions for each 2-min period. This careful control is needed to assure that any observations of tracer concentration downstream result from hydrologic change and not from pump variation.

Synoptic samples were collected at 33 stream and 22 inflow locations on September 16 after the bromide concentration reached a steady-state plateau. Sampled inflows included tributary flows at Wildcat Gulch (1,265 m), Peruvian Gulch (3,047 m), Gad Valley Gulch (4,480 m), White Pine Fork (7,250 m), and a stream through the Tanners Flat campground (8,053 m). Inflow from mines or mine tunnels was sampled at the combined inflow of the Columbus–Rexall and Howland tunnels (CCR at 1,403 m), the Superior Point mine (2,470 m), the discharge of

cooling water from the Snowbird resort co-generation plant (2,926 m), and the WDT discharge (WDT at 3,068 m). The remaining inflows were mostly various sized springs.

Samples were collected in 1.8-L HPDE bottles usually by submersing the neck of each bottle into the water near the thalweg. Where the stream was deep enough and wide enough, samples were collected by integrating techniques (Ward and Harr 1990). Samples were transported to a central processing area where 125-mL aliquots were prepared for cation and anion analyses. Onsite processing included filtration, pH measurement, and preservation of samples for iron speciation. Filtration was completed with in-line disk filters with 0.45- μ m membranes. Aliquots for iron speciation were placed in amber bottles and preserved with concentrated hydrochloric acid to fix the ferrous/ferric ratio in filtered samples (To et al. 1998). Ultrafiltration was completed for four stream sites (1,464, 3,037, 7,315, and 8,300 m) using tangential-flow filtration with 10,000-Da molecular weight membranes. As in other studies of mine drainage, this was to evaluate the truly dissolved concentrations of metals (Kimball et al. 1992, 1995). Aliquots for cation analysis were acidified to pH <2.0 with ultrapure nitric acid. Total recoverable and dissolved cation concentrations were determined from unfiltered and filtered samples, respectively, using inductively coupled argon plasma-mass spectrometry. Cation concentrations are reported for Al, As, Ba, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Mo, Na, Ni, Pb, Si, Sr, and Zn. Dissolved anion concentrations were determined from filtered, unacidified samples by ion chromatography. Anion concentrations are reported for Cl, Br, and SO₄. Ferrous (FeII) and total dissolved iron (FeT) were determined colorimetrically with a method modified from Brown et al. (1970). Alkalinity (as calcium carbonate) was determined from filtered, unacidified samples by titration.

In an effort to reduce study costs, only a subset of the samples collected in 2004 were subject to the complete cation and anion analyses described above. For the case of stream samples, complete cation and anion analyses for dissolved and total recoverable aliquots were obtained for 17 of the 33 sampled sites. For the case of inflow samples, complete cation and anion analyses for dissolved aliquots were obtained for 9 of the 22 sampled sites. Chemical analyses for the remaining 16 stream and 13 inflow sites included only pH, anion concentrations (F and SO₄), and zinc concentration (dissolved and total recoverable). With the exception of zinc, none of the 2004 inflow samples were analyzed for total recoverable cation concentrations. Due to the incomplete analysis of the 2004 samples, additional sources of data are used in the modeling analysis of remedial options that follows. These additional data sources include the 1998 study (Gerner et al. 2001; Kimball et al. 2001) and unpublished data from August 1999.

Simulation of existing conditions

The first step in evaluating remedial options for LCC is to characterize and quantify the hydrologic and geochemical processes that affect metal concentrations under existing conditions. To characterize existing conditions, a reactive solute transport model that mechanistically describes the relevant processes was calibrated. The calibrated model can then be modified to simulate proposed remedial actions, and estimate post-remediation water quality. The reactive solute transport model used for this purpose is OTEQ (Runkel et al. 1996a, 1996b), a model formed by coupling the OTIS solute transport model (Runkel 1998) with a chemical equilibrium submodel. The equilibrium submodel is based on MINTEQA2 (Allison et al. 1991), a model that calculates the distribution of chemical species that exist within a batch reactor at equilibrium. The coupled model considers a variety of processes including advection, dispersion, transient storage, transport, and deposition of water-borne solid phases, acid-base reactions, complexation, precipitation-dissolution, and sorption. Governing equations are formulated in terms of chemical components, where the total component concentration is the sum of all dissolved, precipitated, and sorbed species. Precipitated and sorbed species may reside within the water column or on the streambed; precipitated and sorbed species within the water column settle in accordance with a specified settling velocity and depth. Total component concentrations are partitioned between dissolved, precipitated, and sorbed phases based on equilibrium submodel calculations for each model segment. Components used in the LCC application include Al, CO₃ (total inorganic carbon), Ca, Cd, Cu, F, Fe(III), Mg, Mn, SO₄, TOH (total excess hydrogen), and Zn. Development of a calibrated model of existing conditions requires (1) quantification of hydrologic parameters, (2) specification of geochemical reactions and equilibrium constants, and (3) assignment of boundary conditions and inflow chemistry.

Quantification of hydrologic parameters

Hydrologic parameters within OTEQ include streamflow, main channel cross-sectional area, settling velocity, and settling depth. Spatial variation in these hydrologic parameters was considered by breaking the study reach into 28 model reaches based on changes in streamflow (Table 1). Estimates of streamflow were developed using the tracer dilution method (Kilpatrick and Cobb 1985) and the observed dilution of the bromide tracer. Tracer breakthrough curves at 1,260 and 8,300 m were used to calculate an average velocity (0.18 m/s) that was used with streamflow estimates to calculate the main channel cross-sectional area for each model reach. Settling velocity for

Table 1 Model reaches including streamflow and inflow locations

Reach and distances (m)	Streamflow (L/s)		Locations of observed inflows (m)	Inflow assignment (year-inflow: elements)
	Top of reach	Within reach increase		
1: 1151–1260	24.2	0.0	None	NA
2: 1260–1340	24.2	8.3	1265-Collins Gulch	98-1265 T:Al, Cd, Cu, Mn; D:Ca, Mg 04-1265 T:Zn See Table 2:FeIII
3: 1340–1413	32.5	11.3	1403-Columbus-Rexall (CCR)	04-1403 D:Zn See Table 2: Al, Ca, Cd, Cu, Fe(III), Mg, Mn
4: 1413–1464	43.8	.0	None	NA
5: 1464–1742	43.8	1.2	1490, 1592	98-1490 T:Al, Cu; D: Ca, Mg 98-1592 T: Mn 04-1490 D: SO ₄ 04-1592 D: F See Table 2: Cd, Zn, Fe(III)
6: 1742–1848	45.0	.0	None	NA
7: 1848–1951	45.0	1.5	1874-Hellgate Spring	98-1874 D: Ca, Fe(III), Mg, Mn; T: Cu See Table 2: Al, Cd, Zn
8: 1951–2255	46.5	3.5	None	98-1874 D: Ca, Fe(III), Mg, Mn; T: Cu 4-1874 D: F, SO ₄ See Table 2: Al, Cd, Zn
9: 2255–2501	50.0	3.4	2416, 2470-Superior Point mine	04-2470 D: Ca, Cd, Cu, F, Fe(III), Mg, Mn, SO ₄ See Table 2: Al, Zn
10: 2501–2618	53.3	2.3	None	04-2470 D: Ca, Cu, F, Fe(III), Mg, Mn, SO ₄ See Table 2: Al, Cd, Zn
11: 2618–2719	55.6	2.0	2666	99-2666 D: Ca, Cu, Fe(III), Mg, Mn See Table 2: Al, Cd, Zn
12: 2719–2914	57.7	4.2	2892	98-2892 T: Fe(III); D: Al, Ca, Cu, Mg, Mn See Table 2: Cd, Zn
13: 2914–3037	61.9	2.5	2925, 2926-Cogen	04-2926 D: Ca, Cd, Cu, F, Fe(III), Mg, Mn, SO ₄ , Zn 98:2926 D: Al
14: 3037–3065	64.4	2.7	3047-Peruvian Gulch	04-3047 D: Ca, Cd, Cu, Fe(III), Mg, Mn, Zn 98-3047 T: Al
15: 3065–3118	67.1	72.1	3068-WDT, 3069	04-3068 D: Ca, Cd, Cu, F, Mg, Mn, SO ₄ , Zn 98-3068 T: Al See Table 2: Fe(III)
16: 3118–3245	139.2	.8	3125	98-3125 T: Al, Cd, Cu, Fe(III), Mn; D: Ca, Mg See Table 2: Zn
17: 3245–3494	140.0	1.7	3351	98-3351 T: Al, Fe(III), Mn; D: Ca, Cd, Cu, Mg See Table 2: Zn
18: 3494–3696	141.7	1.4	3590, 3671	98-3671 T: Al, Cu, Fe(III); D: Ca, Cd, Mg, Mn 04-3671 D: SO ₄ , F See Table 2: Zn
19: 3696–4084	143.1	.0	None	NA
20: 4084–4303	143.1	4.3	Non	Same as reach 21

Table 1 continued

Reach and distances (m)	Streamflow (L/s)		Locations of observed inflows (m)	Inflow assignment (year-inflow: elements)
	Top of reach	Within reach increase		
21: 4303–4540	147.4	4.0	4480-Gad Valley Gulch	04-4480 D: Ca, Cd, Cu, Fe, Mg, Mn 04-7250 D: Al See Table 2: Zn
22: 4540–6056	151.4	26.2	None	Same as reach 21
23: 6056–6775	177.6	6.4	None	Same as reach 21
24: 6775–7234	184.1	7.4	None	Same as reach 21
25: 7234–7315	191.4	43.2	7250-White Pine Fork	04-7250 D: Al, Ca, Cd, Cu, Fe(III), Mg, Mn; T: Zn
26: 7315–7592	234.6	10.5	None	Same as reach 25
27: 7592–8009	245.1	27.8	7787	Same as reach 28
28: 8009–8300	272.9	51.9	8053	04-8053 D: Al, Ca, Cd, Cu, Fe(III), Mg, Mn, Zn

Unless noted, F and SO₄ concentrations are set using the inflow's dissolved concentration from 2004; Inflow assignment: year-inflow sample location; T total concentration, D dissolved concentration, NA not applicable, locations with complete 2004 dissolved concentrations shown in bold

precipitated and sorbed species in the water column was set to 2×10^{-5} m/s (Broshears et al. 1996). Settling depth was specified for each model reach and adjusted by trial-and-error to obtain a close correspondence between the simulated and observed total recoverable concentration for Fe(III).

Specification of geochemical reactions and equilibrium constants

As with other OTEQ applications, the geochemical reactions are thought to be sufficiently fast such that the assumption of chemical equilibrium applies. Unless noted otherwise, equilibrium constants for all acid/base, complexation, precipitation, and sorption reactions were set using default values from the equilibrium submodel. These default values are based on a revised version of the MINTEQ database (Allison et al. 1991; Dzombak and Morel 1990) that is consistent with the WATEQ4F database distributed with PHREEQC (Parkhurst and Appelo 1999). Equilibrium constants and activity coefficients were adjusted for the effects of temperature (9°C) and ionic strength (0.0045 M) within the equilibrium submodel. Precipitation reactions for Al and Fe(III) were defined using microcrystalline gibbsite [Al(OH)₃, default log $k = -8.77$] and ferrihydrite [Fe(OH)₃, default log $k = -4.89$] as the solid phases.

Sorption of Ca, Cd, Cu, SO₄, TOT, and Zn to freshly precipitated Fe oxides was modeled using a surface complexation approach and the database of Dzombak and Morel (1990, see also Allison et al. 1991; Runkel et al. 1999). The mass of sorbent within each model segment was based on the amount of precipitated Fe(III) within the

water column, as determined by the equilibrium submodel. Precipitated Fe(III) on the streambed was assumed to be saturated with respect to sorbed species and is therefore not a sink in the steady-state analysis presented here. Sorbent molecular weight and specific surface area were set using the best estimates of Dzombak and Morel (1990). The low- and high-affinity site densities for the sorbent phase were set at 0.3, which is the upper value reported by Dzombak and Morel (1990), and 0.058 moles of sites per mole of sorbent. The surface complexation constant for sorption of copper onto high-affinity sites (default log $k = 2.89$) was set to the upper value of log $k = 3.4$, reported by Dzombak and Morel (1990, see also Runkel et al. 1999).

Inflows of dissolved inorganic carbon (CO₃) result in degassing when the stream becomes oversaturated with respect to atmospheric CO₂. Degassing was modeled as an equilibrium process by specifying the partial pressure of atmospheric CO₂, adjusted for elevation. The corresponding log k value for LCC degassing is equal to 21.8. Initial simulations using this calculated log k value resulted in simulations of pH that exceeded observed pH along the entire length of Little Cottonwood. The log k value was subsequently adjusted downward (to 21.5) to improve the correspondence between simulated and observed pH.

Assignment of boundary conditions and inflow chemistry

A key part of the calibration process specifies component concentrations at the upstream boundary of the modeled system (the first site below the injection at 1,151 m) and within the inflow waters entering each reach. This task was especially difficult for the LCC application due to the

incomplete chemical analyses of the 2004 samples noted above. Unless noted otherwise, all references to data and samples discussed below pertain to the 2004 data set; use of 1998–1999 data is noted where appropriate.

Upstream boundary concentrations for Al, Ca, Cd, Cu, and Fe(III) were set using the total recoverable concentrations observed at 1,260 m, the site closest to the upstream boundary with complete cation and anion chemistry. Boundary concentrations for F, Mg, Mn, and SO₄ were set using the dissolved concentrations observed at 1,151 m; the Zn boundary concentration was set using the total recoverable concentration at 1,151 m.

Most reaches include one or more sampled inflows that were used to set component inflow concentrations (Table 1). When more than one inflow was available for a given reach, the largest observed inflow was generally used. Component inflow concentrations for reaches without observed inflows were set using inflow data from a nearby location (e.g., data from White Pine Fork were used to specify concentrations in reach 26; Table 1). The type (dissolved vs. total recoverable) and source (1998–1999 vs. 2004) of inflow data used varied from reach to reach. As noted previously, nine of the inflows sampled in 2004 have complete cation and anion analyses for dissolved concentration. Component inflow concentrations for reaches that use these inflows were therefore set using the 2004 dissolved concentrations, where possible (reaches 9–10, 13–15, 20–28, Table 1). Inflow concentrations for most components in reaches that use one of the remaining 13 inflows were set using dissolved or total recoverable concentrations from 1998 or 1999; inflow concentrations for SO₄, F, and Zn in these reaches were set using the available 2004 data (e.g., reach 2, Table 1).

Incomplete analyses of the 2004 inflow samples resulted in a difficult inflow assignment process that sometimes deviated from the general description provided above (Tables 1, 2). Use of total recoverable values to set inflow concentrations is generally preferred over the use of dissolved values when the inflows of interest flow directly into the stream. Use of total recoverable concentrations in reaches that rely exclusively on 2004 data was not possible, however, as only dissolved data were available. This shortcoming of the available data did not pose a problem in most reaches, where the use of dissolved concentrations appears to reproduce the observed instream concentrations downstream of the inflows. One exception to this general statement is reach 3, where the use of dissolved concentrations from the 2004 sampling of the Columbus–Rexall results in a gross underestimation of the instream concentration downstream of the inflow. Unlike the other inflows (e.g., Al in reach 15, Table 1), use of total recoverable data from 1998–1999 is not appropriate given the change in conditions (recall that the fen treatment was on in 1998–

1999 and off in 2004). Component inflow concentrations in reach 3 were therefore set such that the observed total recoverable concentrations were reproduced by the simulation at the stream site downstream of the Columbus–Rexall (e.g., Figs. 3a, 4a). Additional deviations from the general inflow assignment process are detailed in Tables 1 and 2.

Concentrations at the upstream boundary and within the inflows were set equal to observed concentrations for most components, as described above. Two exceptions are TOTH and CO₃, components were assigned concentrations based on stand-alone MINTEQ computations. In these computations, pH and alkalinity were fixed at observed values from 2004, and TOTH and CO₃ were determined from the equilibrium speciation. For samples without alkalinity, CO₃ concentrations were based on equilibrium with atmospheric CO₂.

Simulation of remedial options

Remediation was modeled by running the calibrated model of existing conditions with varying the level of treatment by the fen (CCR inflow) and the level of discharge from the WDT. Results of each variation provide concentrations that can be compared to water-quality standards to allow for the evaluation of hypothetical plans. The difference in concentration between samples of the CCR inflow in 1998 and 2004 indicates the magnitude of treatment by the fen in 1998 (Table 3). Mine-related constituents (Cd, Cu, Zn) had higher concentrations in 2004 than in 1998 and the effect on concentrations in the stream also was greater. Thus, the inflow chemistry from 2004 was considered the no treatment option for the simulations (labeled Fen04), the 1998 chemistry was considered the proven extent of fen treatment (Fen98), and two further treatment options considered additional levels of reduction in all constituents. Fen98b considered an additional 20% reduction from the Fen98 concentrations for all components (excluding TOTH, pH is equal to the observed 1998 value), and Fen98c considered an additional 29% decrease beyond Fen98b. For example, the inflow concentration of Zn was 2,450 µg/L for the untreated option (Fen04), 442 µg/L for the treated option at the 1998 level (Fen98), 350 µg/L for the first additional degree of fen treatment (Fen98b), and 250 µg/L for the greatest extent of fen treatment (Fen98c).

Treatment options for the WDT consisted of various percentages of the 2004 inflow. The 2004 level of inflow was the maximum load contribution from the WDT because it resulted from draining the mine pool behind the bulkhead of the tunnel at the greatest rate possible (labeled WT100). Three other options were simulated at 75, 50, and 25% of that inflow (WT75, WT50, and WT25). The inflow chemistry was the same for each of these levels, but the

Table 2 Calibration inflow concentrations that are not equal to observed inflow data

Reach	Component	Calibration value ($\mu\text{g/L}$)	Comparable data		Notes
			Value ($\mu\text{g/L}$)	Data source	
2	Fe(III)	3000	320	98-1265 T	See text discussion
3	Al	1,150	6	04-1403 D	See text discussion
	Ca	56,994	34,966	04-1403 D	See text discussion
	Cd	13	16	04-1403 D	See text discussion
	Cu	597	130	04-1403 D	See text discussion
	F	190	30	04-1403 D	See text discussion
	Fe(III)	6,998	1	04-1403 D	See text discussion
	Mg	15,833	10,992	04-1403 D	See text discussion
	Mn	430	560	04-1403 D	See text discussion
	SO ₄	96,742	71,050	04-1403 D	See text discussion
5	Cd	0	1	98-1490 T	Concentration reduced to improve simulation
	Fe(III)	1	3	98-1490 T	Concentration reduced to improve simulation
	Zn	0	40	04-1490 T	Concentration reduced to improve simulation
7	Al	100	30	98-1874 T	Concentration reduced to improve simulation
	Cd	2	9	98-1874 D	25% of 98-1874 concentration
	Zn	63	250	04-1874 T	25% of 98-1874 concentration
8	Al	100			Concentration increased to improve simulation
	Cd	0			Concentration reduced to improved simulation
	Zn	0			Concentration reduced to improved simulation
9	Al	100	9	04-2470 D	Concentration increased to improve simulation
	Zn	79	316	04-2470 D	25% of 04-2479 concentration
10	Al	100			Concentration increased to improve simulation
	Cd	0			Concentration reduced to improved simulation
	Zn	0			Concentration reduced to improved simulation
11	Al	100	184	99-2666 T	Concentration increased to improve simulation
	Cd	0	2	99-2666 T	Concentration reduced to improved simulation
	Zn	0	276	99-2666 T	Concentration reduced to improved simulation
12	Cd	0	1	98-2892 T	Concentration reduced to improved simulation
	Zn	0	32	04-2892 T	Concentration reduced to improved simulation
15	Fe(III)	30	2	04-3068 D	From mass balance of total recoverable stream concentrations
16	Zn	0	85	04-3125 T	Concentration reduced to improved simulation
17	Zn	0	37	04-3551 T	Concentration reduced to improved simulation
18	Zn	0	29	04-3671 T	Concentration reduced to improved simulation
20–24	Zn	0	11	04-4480 D	Concentration reduced to improved simulation

$\mu\text{g/L}$ micrograms per liter, *T* total concentration, *D* dissolved concentration

discharge was decreased to simulate a smaller load for each option. These levels of loading could represent a decrease in the operational release from the tunnel or else a dilution of the inflow by snowmelt runoff.

Sixteen combinations of the CCR and WDT options were simulated. Each combination was simulated with and without the effects of settling, and the reported results represent the worst case for each simulation (settling vs. no

settling). Modeling results are reported as a percent exceedence value in the form:

$$\text{Percent exceedence} = 100 \left(\frac{C_m - C_s}{C_s} \right), \quad (1)$$

where C_m is the modeled concentration, in mg/L, and C_s is the concentration of the instream standard for the given hardness from the model, in mg/L.

Table 3 Comparison of inflow and instream concentrations between 1998 and 2004, downstream from the combined Howland–Columbus–Rexall discharge

Constituent	1998-Fen on		Downstream comparison	2004-Fen off	
	CCR inflow, dissolved	Downstream site, total		CCR inflow, dissolved	Downstream site, total
Aluminum	20	28	<	6	88
Cadmium	5	2.0	<	16	3.5
Copper	30	21	<	131	125
Iron	<5	92	≪	.6	540
Zinc	407	116	≪	2,400	524
pH	8.39	8.29	>	7.48	7.57

All concentrations in micrograms per liter; CCR combined Howland–Columbus–Rexall inflow, < less, > greater than

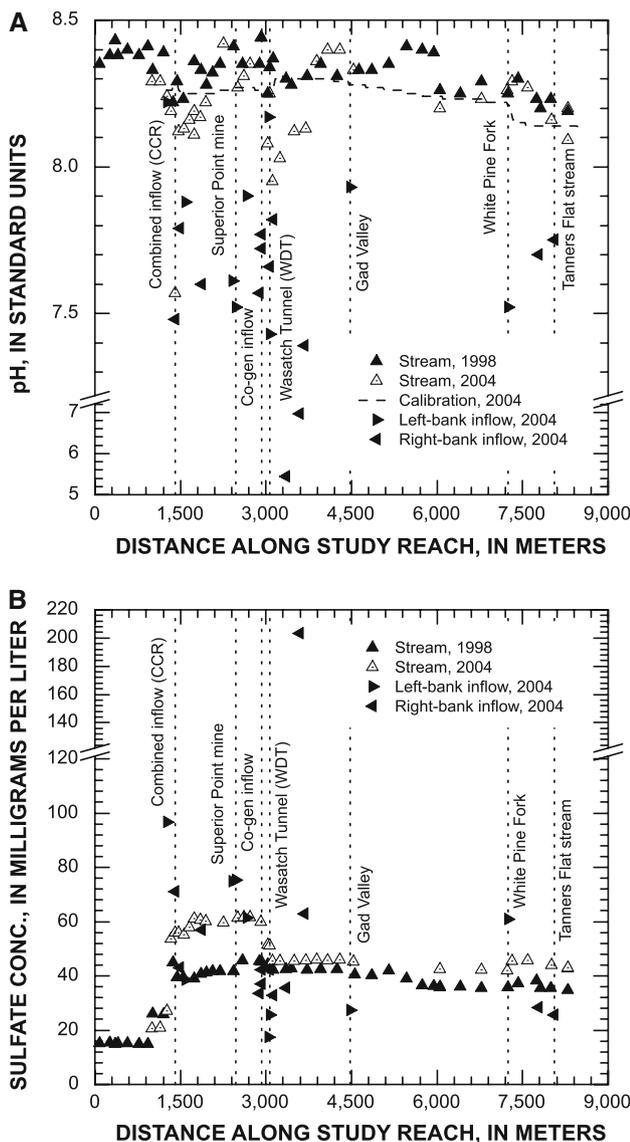


Fig. 2 Variation of (a) measured and simulated pH and (b) measured sulfate with distance along the study reach, Little Cottonwood Creek, Utah

Acute and chronic water-quality standards come from the Utah Department of Administrative Services (2005). A positive value for the percent exceedance indicates that the water-quality standard is not met by the simulation, and a negative value indicates the standard is met.

Results and discussion

Low-flow water quality, 1998 and 2004

Characterization of instream solute concentrations indicates particular geochemical processes that should be simulated by the calibrated model. Results of chemical characterization for synoptic samples from the 2004 study are listed in the Electronic Supplement (Tables E2, E3) Two principal distinctions between the 1998 and 2004 synoptic data sets are illustrated by variations in pH and SO₄ (Fig. 2). Values of pH for most samples fell into a narrow range from about 8.0 to 8.5 (Fig. 2a), but a noticeable decrease in pH occurred downstream from both the CCR and WDT inflows in 2004. Higher SO₄ concentration occurred downstream from the CCR inflow in 2004 than in 1998, but SO₄ concentration was quite similar at other locations along the study reach in both years (Fig. 2b). Both the lower pH and the higher SO₄ concentration resulted from the lack of treatment by the fen in 2004. One additional distinction for SO₄ concentration occurred at the Superior Point mine (Fig. 2b), where SO₄ concentration increased more in 1998 than in 2004. As part of the 2004 study, the mine pool level intentionally had been lowered to substantially decrease the flow from the Superior Point mine.

Higher concentrations of metals also resulted from the lack of treatment during the 2004 synoptic study. For example, the high concentration of Fe from the CCR discharge mixed into the high pH water of LCC and resulted in the rapid formation of Fe-rich colloids (as indicated by

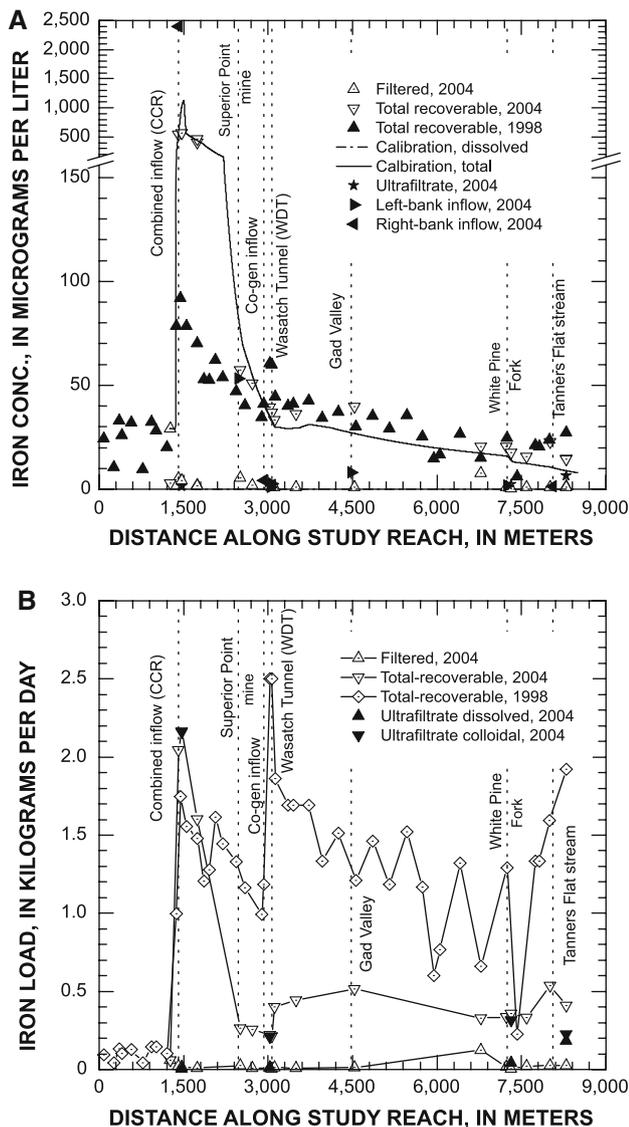


Fig. 3 Variation of (a) measured and calibrated iron concentration, and (b) dissolved and total recoverable iron load with distance along the study reach, Little Cottonwood Creek, Utah

the substantial difference between total recoverable and filtered concentrations, Fig. 3a). The ultrafiltrate sample indicates an even greater colloidal Fe load; the 0.45- μm filtration allows a portion of the colloidal Fe to pass through and be measured as “dissolved” Fe rather than colloidal Fe (Kimball et al. 1992, 1995). The total recoverable concentration, which essentially represents the colloidal concentration, decreased downstream from the CCR inflow, and the decrease of Fe load confirms that this represents settling of Fe colloids from the stream in the reach from 1,420 to 2,155 m (Fig. 3b). The greater total recoverable load of Fe in 1998 than in 2004 most likely represents the decrease in load from the substantial drought conditions that preceded the 2004 synoptic study

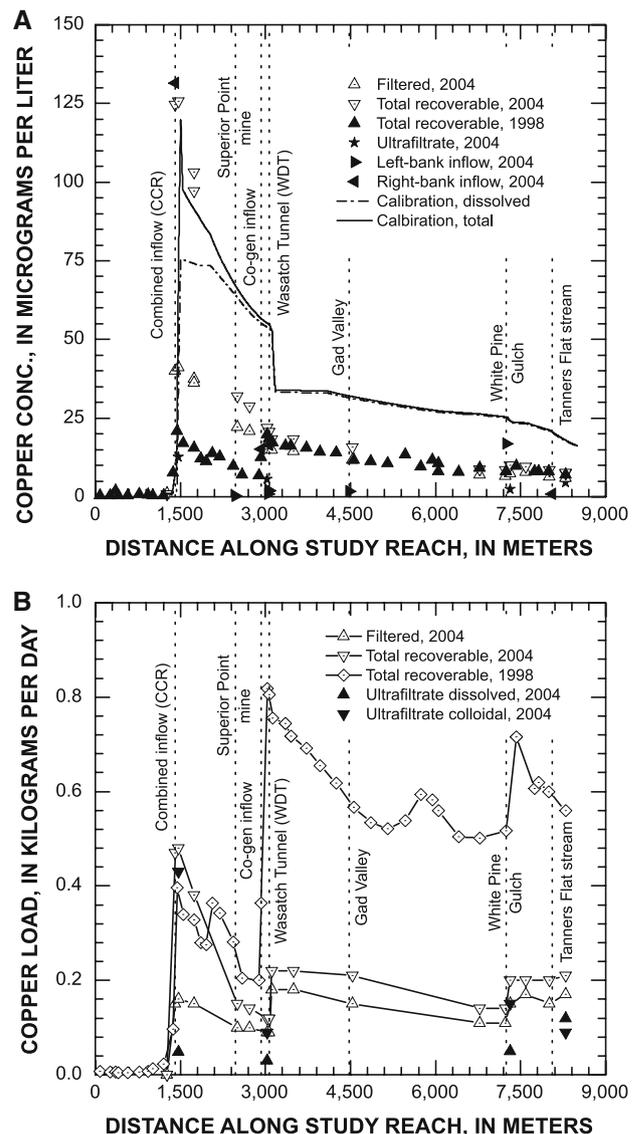


Fig. 4 Variation of (a) measured and calibrated copper concentration, and (b) dissolved and total recoverable copper load with distance along the study reach, Little Cottonwood Creek, Utah

(Wilkowske et al. 2003). Aluminum concentration indicated that colloidal Al also formed rapidly in the mixing zone downstream from the CCR inflow, and colloidal Al settled in the same reach (Table E3 of the Electronic Supplement).

The formation and settling of Fe colloids had an important influence on Cu and Zn concentrations (Figs. 4, 5). The 2004 data indicate that Cu principally occurred in the colloidal phase downstream from the CCR inflow where a substantial increase in Cu concentration occurred (Fig. 4a). Partitioning of Cu to the colloidal phase at a pH greater than 8.0 likely resulted from sorption of Cu to the colloids (Runkel et al. 1999; Smith 1999). The 0.45- μm filtered sample indicated that 33% of the Cu occurred in the

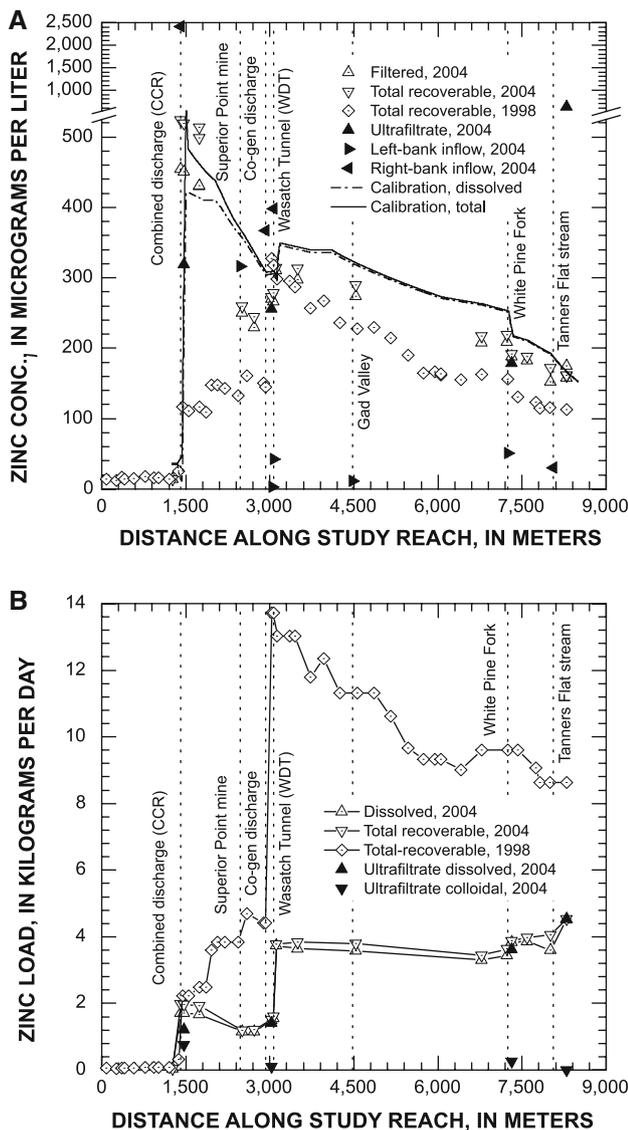


Fig. 5 Variation of (a) measured and calibrated zinc concentration, and (b) dissolved and total recoverable zinc load with distance along the study reach, Little Cottonwood Creek, Utah

colloidal phase, but the ultrafiltrate sample indicated that 90% of the Cu was in the colloidal phase. Copper load followed the same pattern as Fe load, suggesting that the colloidal Fe settling strongly affected the fate of Cu (Fig. 4b). Concentrations of Zn downstream from the CCR inflow also indicated sorption to Fe colloids (Fig. 5a), but not to the extent of Cu sorption. At 1,464 m, where an ultrafiltrate sample was obtained, the 0.45-µm filtration indicated that 13% of the Zn was in the colloidal phase. Ultrafiltration, however, indicates that as much as 38% of the Zn was in the colloidal phase (Table E3 of the Electronic Supplement). Again, following the pattern of Fe load, the Zn load decreased from 1.71 to 1.15 kg/day between 1,410 and 2,501 m (Fig. 5b). Zinc load from the

CCR inflow was substantial, but the greatest Zn load was contributed by the WDT inflow, both in 1998 and in 2004. Thus, concentrations of Cu and Zn were affected by sorption to Fe colloids and subsequent settling of colloids from the water column.

Simulating existing conditions

Existing conditions were modeled using the 2004 data set to indicate no flow through the fen and 100% flow from the WDT. Results are reported for three locations where compliance was studied as part of the TMDL, including (1) downstream from the CCR inflow (1,567 m), (2) downstream from the WDT (3,494 m), and (3) at the U.S. Geological Survey gaging station, 10167450, LCC at Tanner Flat campground near Alta, Utah (8,300 m, Fig. 1). Modeling results for these conditions are illustrated for pH, Fe(III), Cu, and Zn.

The simulation of pH matches the narrow range of sampled pH along the study reach, but does not match the decreases in pH that occurred downstream from both the CCR and WDT inflows (Fig. 2a). At these high-modeled pH values, results for Fe(III) are consistent with the tendency for Fe to be completely in the colloidal phase and reasonably follow the sampled pattern of observed Fe(III). Model simulations of dissolved Fe(III) concentration are essentially less than detection (Fig. 3a). Simulation results for Fe(III) are important for simulation of Cu and Zn sorption because of Fe colloids in the water column (Fig. 3b). For both Cu and Zn, the partitioning to the colloidal phase by sorption onto precipitated Fe(III) is simulated downstream from the CCR inflow (Fig. 4), but total recoverable concentrations of both metals are slightly overestimated by the model. The simulated amount of Zn sorption is in general agreement with observed data (Fig. 5b), whereas the amount of Cu sorption is underestimated (simulation > dissolved data, Fig. 4a). Simulations of Ca and Mg (not shown), two components used to determine hardness-based water-quality standards, reproduce the general features of the observed concentration profiles, but were slightly less than the observed concentrations along the entire study reach.

The simulation results discussed above are the product of an unusually difficult calibration process that included a number of assumptions and decisions that warrant further discussion. Simulation of Cu and Zn sorption onto precipitated Fe(III) is especially troublesome as it required extreme values of the high-affinity site density and the Fe(III) inflow concentrations (reaches 2 and 3, Table 2). Previous OTEQ applications utilize a high-affinity site density equal to the upper value reported by Dzombak and Morel (1990, 0.01 moles of sites per mole of sorbent), reflecting the high sorptive capacity of freshly precipitated

Fe oxides (Runkel et al. 1999; Runkel and Kimball 2002). The present application uses a site density that is nearly six times this upper value, well beyond the realistic range for sorption onto Fe(III) precipitates. This result may be explained in part by the fact that simulation results are based on sorption onto Fe(III) precipitates, whereas the observed data may result from the presence of additional sorbents, such as Al precipitates (Anderson and Benjamin 1990) and biofilms (Toner et al. 2005). Although sorption onto these additional sorbents is not explicitly modeled, specification of extreme site densities and Fe(III) inflow concentrations represents an ad hoc way of accounting for their effect.

A related issue is the specification of settling depths for the precipitated and sorbed species. The specified depth upstream of the WDT ranged from 1 to 10 cm, whereas settling depth downstream of the WDT ranged from 80 to 100 cm. Although the LCC is generally shallow upstream of WDT, the specified settling depths are less than observed water depth. This discrepancy may be a result of stream hydrodynamics such as hyporheic flow, which may strain colloids from the water column as water passes through porous areas of the streambed. Settling depths downstream of the WDT, in contrast, generally exceed observed depth. This discrepancy may result from the fact that streamflow more than doubles as the WDT inflow enters Little Cottonwood. The increase in flow is accompanied by higher stream velocities that may act to keep colloidal materials in suspension, such that the effective settling depth exceeds the actual water depth.

Although the difficult calibration process may be explained partially by unmodeled processes (e.g., straining, additional sorbents), several aspects of the 2004 field effort also contribute to the problem. The lack of total recoverable data for the inflow samples collected in 2004 is especially notable, as total recoverable concentrations (rather than dissolved) represent an inflow's contribution to the stream when the incoming water flows directly to the stream. Observed metal concentrations downstream of inflows are the net result of both inflow and reaction. As such, the availability of total recoverable inflow data represents an important constraint on the modeling that allows for the separation of the inflow loading and geochemical reactions. This constraint is sorely lacking in inflows such as CCR (reach 3, Table 2), where considerable loading and reaction take place. A related issue is the lack of complete cation and anion data for both total recoverable and dissolved concentration for many of the inflows (see inflow locations that are not in bold, Table 1). As a result, inflow concentrations for the corresponding reaches were set based on 1998–1999 data, under the assumption that comparable inflow loading occurred in 2004 (Table 2).

Simulation of remedial options

The unusually difficult calibration step discussed above suggests considerable uncertainty that must be addressed when modeling the effects of potential remedial options. One way of considering this uncertainty is to “error on the conservative side”, by producing simulations which underestimate the removal of metals and/or water hardness. Simulated Cu and Zn concentrations from the calibration, for example, underestimate removal (simulated concentrations exceed observed concentrations, Figs. 4, 5). As a result, simulations of remedial options are also likely to underestimate removal, leading to the selection of more stringent remedial actions. Similarly, simulated values of water hardness (Ca and Mg) underestimate hardness, resulting in lower hardness-based water-quality standards. More stringent remedial actions will therefore be needed to meet the standards.

In addition to erring on the conservative side during model calibration, three aspects of the remediation simulations are used to counteract model uncertainty. First, each of the remediation combinations is modeled with and without the effects of settling. Simulation results presented below utilize the worst case (highest dissolved concentration) when calculating the percent exceedence (Equation 1). Second, all component concentrations in the CCR inflow are reduced by an identical percentage for remediation options Fen98b and Fen98c. These reductions for Ca and Mg result in lower water hardness downstream, and lower hardness-based standards. Third, a given remediation combination is only deemed “compliant” if the simulated concentration is at least 10% below the corresponding water-quality standard (i.e., the percent exceedence is –10% or less). This third aspect provides an important “margin of safety” by eliminating the consideration of remedial options that result in concentrations that are very close to the standards. With this background, the results of simulations for all the combinations of remediation options are evaluated with respect to the compliance locations.

Site 1: Downstream from the combined Howland–Columbus–Rexall discharge

At the compliance site downstream from the CCR inflow, in the absence of any effect from the downstream inflow of the WDT, water-quality standards for both Cu (Table 4) and Zn (Table 5) are exceeded when there is no treatment by the fen (Fen04). With the level of treatment from the fen in 1998 (Fen98), concentrations of Cu and Zn do not exceed the acute or chronic standards at this upstream compliance site. In fact, only for the chronic standard of Cu does the percent exceedence even come as close as 10% of the standard. Thus, if the CCR inflow were the only

Table 4 Percent exceedence of copper for simulations of remediation options, Little Cottonwood Creek, Utah

WDT option	CCR option							
	Acute				Chronic			
	Fen 04	Fen 98	Fen 98b	Fen 98c	Fen 04	Fen 98	Fen 98b	Fen 98c
Downstream from CCR								
NA	175	-42	-52	-63	330	-10	-26	-44
Downstream from WDT								
WT100	50	-42	-45	-49	134	-10	-15	-20
WT75	58	-43	-46	-50	147	-11	-17	-23
WT50	68	-44	-49	-53	162	-13	-20	-27
WT25	79	-47	-52	-58	179	-18	-26	-35
Downstream from Tanners Flat								
WT100	9	-48	-50	-52	65	-22	-25	-28
WT75	11	-49	-51	-53	67	-24	-27	-30
WT50	12	-50	-52	-54	68	-25	-28	-32
WT25	13	-51	-54	-56	69	-27	-31	-35

WDT Wasatch drainage tunnel, CCR combined Howland–Columbus–Rexall discharge, italicized values indicate compliance with the water-quality standards

Table 5 Percent exceedence of zinc for simulations of remediation options, Little Cottonwood Creek, Utah

WDT option	CCR option							
	Acute				Chronic			
	Fen 04	Fen 98	Fen 98b	Fen 98c	Fen 04	Fen 98	Fen 98b	Fen 98c
Downstream from CCR								
NA	143	-38	-48	-60	169	-31	-42	-55
Downstream from WDT								
WT100	101	45	43	41	121	60	58	56
WT75	96	32	30	28	117	46	43	41
WT50	91	15	12	9	110	27	23	20
WT25	82	-10	-14	-19	101	-1	-6	-11
Downstream from Tanners Flat								
WT100	37	1	0	-2	51	11	10	8
WT75	30	-10	-12	-13	43	-1	-2	-4
WT50	22	-23	-25	-27	35	-15	-17	-19
WT25	13	-38	-41	-43	25	-32	-35	-37

WDT Wasatch drainage tunnel, CCR combined Howland–Columbus–Rexall discharge, italicized values indicate compliance with the water-quality standards

concern for Cu and Zn loading, the 1998 level of fen treatment would be sufficient to meet remediation goals.

Site 2: Downstream from Wasatch drainage tunnel

The conditions of meeting standards are more dynamic downstream from the WDT inflow. Similar to conditions at the compliance location downstream from the CCR inflow, some level of treatment by the fen is required to meet the water-quality standards, regardless of the level of WDT

inflow (Tables 4, 5; Fen04 columns). For Cu treatment, the Fen98 level of treatment is sufficient at every level of WDT discharge (Table 4). For Zn treatment, however, a decrease in the discharge of the WDT to WT25 is required to meet the acute and chronic standards. For chronic standards of Zn, the level of treatment by the fen needs to be near the Fen98c level to have a margin of exceedence greater than 10%. Thus, downstream from both major contributions of Cu and Zn load, combinations of both treatments are needed to meet the remediation goals.

Site 3: Downstream from Tanners Flat

At the end of the study reach, the concentrations of both Cu and Zn were lower, both as a result of chemical reactions and of dilution from inflows that had low metal concentrations. The only inflows that contribute to the Cu and Zn loads are White Pine Fork (mostly for Cu, Fig. 4a), and the stream draining Tanners Flat campground (mostly for Zn, Fig. 5a). These contributions, however, were not sufficient to cause an increase that exceeded the standards. With the WDT at 75% discharge (WT75), the percent exceedence for the chronic Zn standard had no margin of safety, and at the WT100 level, the chronic standard was exceeded. The level at WT50 was required to meet the chronic Zn standard and to have a margin of safety.

Reactive transport modeling

Evaluation of remedial options for LCC has several implications for reactive transport modeling of metals in surface waters. Previous efforts to model sorption have primarily focused on low-pH environments where colloidal iron is the primary sorbent (Ball et al. 2004; Runkel et al. 1999; Runkel and Kimball 2002, 2007). The present application, in contrast, involves a circum-neutral stream in which significant quantities of both Fe and Al colloids are present. Given the difficulties in modeling Cu and Zn sorption described herein, development of modeling techniques that consider multiple sorbent phases may be warranted (Anderson and Benjamin 1990).

Modeling of circum-neutral waters also introduces a need to consider CO₂ degassing, an important process affecting instream pH. Degassing in the LCC was initially modeled as an equilibrium process, using a log *k* value calculated from the elevation-corrected partial pressure of CO₂. This initial modeling produced a simulated pH that was in excess of observed pH, a discrepancy that can be attributed to the equilibrium approach used within OTEQ. Under the equilibrium approach, degassing occurs instantaneously whenever the water column is oversaturated with respect to the specified partial pressure of CO₂. This approach is inconsistent with theoretical and empirical studies that have shown degassing to be a kinetic process that is limited by transfer across the air/water interface (Choi et al. 2000). The present application mimics this kinetic limitation by utilizing a log *k* value that is lower than the log *k* value associated with the atmospheric partial pressure. Although this change in the log *k* value results in a simulation that reproduces the general features of the observed pH profile, the observed decrease in pH below the key inflows (CCR and WDT) is not reproduced (Fig. 2). Efforts to more accurately simulate pH in circum-neutral waters may therefore require changes in model structure,

such as the addition of kinetic degassing terms to the transport equations for CO₃ and TOTH.

Conclusions

Field-scale experiments provide the means to evaluate remediation options as they are combined with reactive solute transport modeling. In this paper, field data from LCC, Utah, are used to calibrate a reactive solute transport model that considers the hydrologic and geochemical processes affecting metals. The calibrated model is subsequently used to consider various remediation options that are designed to lower Cu and Zn concentrations in the creek, with the goal of meeting water-quality standards at three key monitoring locations. Potential remediation options include treatment of CCR water by a fen, and regulation of discharge from the WDT. Study results have specific implications for LCC, and some general implications for data collection and reactive transport modeling, as summarized below.

Simulation results presented herein suggest that the choice of remedial options will be largely driven by the water-quality standards for dissolved Zn, rather than dissolved Cu (if a remedial option meets the Zn standard, it will also meet the Cu standard). Results of model calibration suggest very little instream removal of Zn under existing, low-flow conditions, such that remedial efforts should focus on reduction at the source. Simulations of remedial options suggest that although regulation of discharge from the WDT results in lower Cu and Zn concentrations, some treatment of the CCR inflow by the fen will be required to meet applicable water-quality standards. Simulation results also suggest that water-quality standards in the upper and lower sections of the study reach (sites 1 and 3) may be met using various combinations of fen treatment and WDT regulation, but that complete compliance in the middle section (site 2) will require the highest level of fen treatment and the largest regulation of WDT discharge (Fen98c and WT25; see chronic standard for Zn, Table 5).

Interpretation of the model results presented here must be conducted in light of model and system uncertainty. Considerable model uncertainty in the LCC application is likely given the unusually difficult calibration process. Much of this uncertainty arises due to the incomplete analysis of the 2004 samples, a measure taken to reduce project costs. This application thus serves to illustrate the trade off between available resources and uncertainty, a trade off that should be considered when conducting future investigations. The effects of model uncertainty are mitigated herein by introducing a margin of safety and by erring on the conservative side during calibration and simulation of remedial options. In addition to the effects of

model uncertainty, decision makers should consider the effects of system uncertainty while evaluating treatment options. Specific concerns include the variability in source chemistry, and variability in treatment effectiveness (e.g., long-term effectiveness of the fen), two issues that are outside the scope of the present study.

Reactive transport modeling is a useful tool for the evaluation of remedial alternatives in complex natural systems, where multiple hydrologic and geochemical processes determine metal fate. Future efforts should recognize the effects of uncertainty and the implications for data collection and analysis. Model development efforts may also be needed to more accurately simulate the processes of sorption and degassing.

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References

Allison JD, Brown DS, Novo-Gradac KJ (1991) MINTEQA2, a geochemical assessment data base and test cases for environmental systems: version 3.0 user’s manual. U.S. EPA, Athens

Anderson PR, Benjamin MM (1990) Modeling adsorption in aluminum-iron binary oxide suspensions. *Environ Sci Technol* 24:1586–1592. doi:10.1021/es00080a020

Ball JW, Runkel RL, Nordstrom DK (2004) Evaluating remedial alternatives for the Alamosa River and Wightman Fork, near the Summitville Mine, Colorado. In: Zannetti P (ed) *Environmental sciences and environmental computing*. The EnviroComp Institute,

Bencala KE, McKnight DM (1987) Identifying in-stream variability: sampling iron in an acidic stream. In: Averett RC, McKnight DM (eds) *Chemical quality of water and the hydrologic cycle*. Lewis Publishers, Inc., Chelsea, MI, pp 255–269

Besser JM, Allert AL, Hardesty DK, May TW, Leib KJ (1999) Seasonal variation in toxicity of streams affected by acid mine drainage. In: 20th Annual Meeting, Philadelphia, Penn., November 14–19, 1999. Society of Environmental Toxicity and Chemistry (SETAC), pp 285

Broshears RE, Runkel RL, Kimball BA, Bencala KE, McKnight DM (1996) Reactive solute transport in an acidic stream: experimental pH increase and simulation of controls on pH, aluminum, and iron. *Environ Sci Technol* 30:3016–3024. doi:10.1021/es960055u

Brown E, Skougstad MW, Fishman MJ (1970) Methods for collection and analysis of water samples for dissolved minerals and gases. U.S. Geological Survey Techniques of Water-Resources Investigation Book 5, Chapter A1

Choi J, Harvey JW, Conklin M (2000) Characterizing multiple timescales of stream and storage zone interaction that affect solute fate and transport in streams. *Water Resour Res* 36:1511–1518. doi:10.1029/2000WR900051

Dzombak DA, Morel FMM (1990) *Surface complexation modeling: hydrous ferric oxide*. Wiley, New York

Germer LJ, Rossi FJ, Kimball BA (2001) Selected hydrologic data for Little Cottonwood Creek, Salt Lake County, Utah, September 1998. U.S. Geological Survey Open-File Report 01-38

Jensen SF, Karimi R (2001) Rocky mountain headwaters mine waste initiative, Alta Wetlands Fen Pilot Project, Interim Report V, 2001 Monitoring Season. Salt Lake County Engineering Division, Salt Lake City

Kilpatrick FA, Cobb ED (1985) Measurement of discharge using tracers. U.S. Geological Survey Techniques of Water-Resources Investigations Book 3, Chapter A16

Kimball BA, McKnight DM, Wetherbee GA, Harnish RA (1992) Mechanisms of iron photoreduction in a metal-rich, acidic stream (St Kevin, Gulch, Colorado, U.S.A.). *Chem Geol* 96:227–239. doi:10.1016/0009-2541(92)90130-W

Kimball BA, Broshears RE, Bencala KE, McKnight DM (1994) Coupling of hydrologic transport and chemical reactions in a stream affected by acid mine drainage. *Environ Sci Technol* 28:2065–2073. doi:10.1021/es00061a014

Kimball BA, Callender E, Axtmann EV (1995) Effects of colloids on metal transport in a river receiving acid mine drainage, upper Arkansas River, Colorado. *USA Appl Geochem* 10:285–306. doi:10.1016/0883-2927(95)00011-8

Kimball BA, Bencala KE, Runkel RL (2000) Quantifying effects of metal loading from mine drainage. In: Fifth International Conference on Acid Rock Drainage, May 13–15, 2000, Denver, Colo. Society for Mining, Metallurgy, and Exploration, Inc., Englewood, Colo., pp 1381–1389

Kimball BA, Runkel RL, Germer LJ (2001) Quantification of mine-drainage inflows to Little Cottonwood Creek, Utah, using a tracer-injection and synoptic-sampling study. *Environ Geol* 40:1390–1404. doi:10.1007/s002540100320

Kimball BA, Runkel RL, Walton-Day K, Bencala KE (2002) Assessment of metal loads in watersheds affected by acid mine drainage by using tracer injection and synoptic sampling: Cement Creek, Colorado, USA. *Appl Geochem* 17:1183–1207. doi:10.1016/S0883-2927(02)00017-3

Kimball BA, Runkel RL, Walton-Day K (2003) Use of field-scale experiments and reactive solute-transport modelling to evaluate remediation alternatives in streams affected by acid mine drainage. In: Jambor JL, Blowes DW, Ritchie AIM (eds) *Environmental aspects of mine wastes*. Mineralogical Association of Canada, British Columbia, pp 261–282

Kimball BA, Johnson KK, Runkel RL, Steiger, JI (2004) Quantification of metal loading to Silver Creek through the Silver Maple Claims area, Park City, Utah, May 2002. U.S. Geological Survey Water-Resources Investigations Report 03-4296

Kimball BA, Walton-Day K, Runkel RL (2007) Quantification of metal loading by tracer injection and synoptic sampling, 1996–2000. In: Church SE, von Guerard PB, and Finger SE (eds) *Integrated investigations of environmental effects of historical mining in the Animas River watershed, San Juan County, Colorado*. U.S. Geological Survey, chap E9, pp 417–495

Leib KJ, Mast MA, Wright WG (2003) Using water-quality profiles to characterize seasonal water quality and loading in the upper Animas River Basin, Southwestern Colorado. U.S. Geological Survey Water-Resources Investigations Report 02-4230

Miller Shepherd (2002) Total maximum daily load for dissolved zinc in the Little Cottonwood Creek. Utah Division of Water Quality, Salt Lake City

Parkhurst DL, Appelo CAJ (1999) User’s guide to PHREEQC (Version 2)—a computer program for speciation, batch-reaction, one-dimensional transport, and inverse geochemical calculations. U.S. Geological Survey Water-Resources Investigation Report 99-4259

- Runkel RL (1998) One-dimensional transport with inflow and storage (OTIS): a solute transport model for streams and rivers. U.S. Geological Survey Water-Resources Investigations 98-4018
- Runkel RL, Kimball BA (2002) Evaluating remedial alternatives for an acid mine drainage stream: application of a reactive transport model. *Environ Sci Technol* 36:1093–1101. doi:[10.1021/es0109794](https://doi.org/10.1021/es0109794)
- Runkel RL, Kimball BA (2007) A simulation-based approach for estimating premining water quality: red Mountain Creek, Colorado. *Appl Geochem* 22:1899–1918. doi:[10.1016/j.apgeochem.2007.03.054](https://doi.org/10.1016/j.apgeochem.2007.03.054)
- Runkel RL, Bencala KE, Broshears RE, Chapra SC (1996a) Reactive solute transport in streams 1, Development of an equilibrium-based model. *Water Resour Res* 32:409–418. doi:[10.1029/95WR03106](https://doi.org/10.1029/95WR03106)
- Runkel RL, McKnight DM, Bencala KE, Chapra SC (1996b) Reactive solute transport in streams 2 Simulation of a pH modification experiment. *Water Resour Res* 32:419–430. doi:[10.1029/95WR03107](https://doi.org/10.1029/95WR03107)
- Runkel RL, Kimball BA, McKnight DM, Bencala KE (1999) Reactive solute transport in streams: a surface complexation approach for trace metal sorption. *Water Resour Res* 35:3829–3840. doi:[10.1029/1999WR900259](https://doi.org/10.1029/1999WR900259)
- Smith KS (1999) Metal sorption on mineral surfaces: an overview with examples relating to mineral deposits. In: Plumlee G, Logsdon MJ (eds) *The environmental geochemistry of mineral deposits part A: processes, techniques, and health issues*. Society of Economic Geologists, Littleton, pp 161–182
- To TB, Nordstrom DK, Cunningham KM, Ball JW, McCleskey RB (1998) A new method for the direct determination of dissolved Fe(III) concentration in acid mine waters. *Environ Sci Technol* 33:807–813. doi:[10.1021/es980684z](https://doi.org/10.1021/es980684z)
- Toner B, Manceau A, Marcus MA, Millet DB, Sposito G (2005) Zinc sorption by a bacterial biofilm. *Environ Sci Technol* 39:8288–8294. doi:[10.1021/es050528+](https://doi.org/10.1021/es050528+)
- Utah Department of Administrative Services (2005) Standards of quality for waters of the state. Division of Administrative Rules, <http://www.rules.utah.gov/publicat/code/r314/r317-002.htm>. Accessed 7 April 2005
- Walton-Day K, Paschke SS, Runkel RL, Kimball BA (2007) Using the OTIS solute transport model to evaluate remediation scenarios in Cement Creek and the upper Animas River. In: Church SE, von Guerard PB, and Finger SE (eds) *Integrated investigations of environmental effects of historical mining in the Animas River watershed, San Juan County, Colorado*. U.S. Geological Survey, chap E24, pp 973–1028
- Ward JR, Harr CA (1990) Methods for collection and processing of surface-water and bed-material samples for physical and chemical analyses. U.S. Geological Survey Open-File Report 89-140
- Wilkowske CD, Allen DV, Phillips JV (2003) Drought conditions in Utah during 1999–2002: a historical perspective. U.S. Geological Survey Fact Sheet 037-03