Long-term Biogeochemistry of Mine Tailings Amended with Organic Carbon for Water-quality Management

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Abstract
Amendment of mine tailings with organic carbon (OC) was evaluated as a technique for water-quality management. The objective of this technique is to limit transport and discharge of sulfide-mineral oxidation products by inducing dissimilatory sulfate reduction (DSR) and metal-sulfide precipitation. Six experimental cells were constructed in the vadose zone of a sulfide- and carbonate-rich tailings deposit characterized by neutral drainage. Varied mixtures of peat, spent brewing grain (SBG) and municipal biosolids (MB) were incorporated into fresh mill tailings. The geochemistry, mineralogy and microbiology of the cells were monitored for six years. Sulfate-reducing conditions developed within three years in cells amended with multiple OC sources. Sulfate removal generally corresponded to alkalinity production, growth of sulfate-reducing bacteria, H_2S production, ^{34}S-SO_4 enrichment and Zn, Mn,and Tl removal. The addition of OC initially induced Fe and As mobilization; however, this process was most pronounced for cells that contained MB. Large increases in pore-water SO_4 concentrations observed immediately below the tailings surface resulted from sulfide-mineral oxidation. Sustained sulfate removal was observed below the oxidation zone in tailings amended with peat+ SBG. Increased SO_4 concentrations observed with other amendments were attributed to declining rates of DSR.

Key Words:sulfate reduction, metal-sulfide precipitation, alkalinity production, passive in situ treatment

Introduction

Drainage from waste rock and tailings deposits can have negative impacts on groundwater and surface water quality at both active and decommissioned mines. This issue is of particular importanceat mines exploiting sulfide-mineral deposits, which are an important source of metals worldwide. The oxidative dissolution of sulfide minerals in mine wastedeposits posesthe primary risk to water quality (Blowes et al., 2003). This process generates H^+ and releases sulfate and metals to pore waters. The dissolution of carbonate minerals can maintain circumneutral pH conditions, however, weakly-hydrolyzing metals (i.e., Zn, Ni, Cu^I, Fe^{II}) and oxyanion-forming metals (i.e., As, Mo, Sb, Se) may remain mobile under these conditions (Heikkinen et al., 2009; Lindsay et al., 2009a).Subsequent acidification due to ongoing sulfide-mineral oxidation and the depletion of acid neutralization capacity can lead to large increases in dissolved metal concentrations(Nordstrom et al., 2000; Moncur et al., 2005; Gunsinger et al., 2006). Unmitigated drainage from sulfide-rich mine waste deposits therefore can have widespread impacts on water quality that persist for decades to centuries (Moncur et al., 2006; Ruiz et al., 2008; Strosnider et al., 2011). Additionally, dissolved organic carbon (DOC), sulfoxyanions and metals contributed with residual mineral processing waters also influence water quality(Smuda et al., 2008; Lindsay et al., 2009a). Managing these potential impacts on water quality during active mining and following closure is fundamental to the environmental sustainability of metal production.

The development of innovative techniques for managing water quality is essential for minimizingenvironmental impacts and financial liabilities associated with mining operations. Substantial

research over recent decades has focused on understanding influences of physical, chemical and biological processes on the geochemical evolution of mine wastes and associated drainage. This research has provided the foundation for important advances in the management of mine water quality. Passive (bio)geochemical techniques developed for source control or migration control are potentially a cost-effective alternative to traditional active treatment technologies, which are generally burdened by high capital, operation and maintenance costs (Blowes, 2002; Johnson and Hallberg, 2005). Source control measures, such as subaqueous tailings disposal or moisture-retaining covers, designed to inhibit sulfide-mineral oxidation may not be practical when active tailings deposition is ongoing.Passive migration-control techniques, including anaerobic wetlands (Hedin et al., 1989; Machemer and Wildeman, 1992), anaerobic bioreactors (Dvorak et al., 1992; Christensen et al., 1996), and permeable reactive barriers (Benner et al., 1997; Jarvis et al., 2006),are commonly employed for remediation of sites impacted by mine drainage. These techniques promote growth of sulfate-reducing bacteria (SRB), which catalyze dissimilatory sulfate reduction (DSR) coupled with organic-carbon oxidation, producing H_2S and carbonate alkalinity. The amendment of mill tailings with organic carbon (OC)has been applied for proactive management of mine drainage quality, through amendment of tailings with OC materials (Hulshof et al., 2006; Lindsay et al., 2009b; Lindsay et al., 2009c; Lindsay et al., 2011). This study evaluates the biogeochemistry of sulfide- and carbonate-rich tailings amended with OCover an extended time period, from 2004 through 2010.

Site Description

The Greens Creek Zn-Ag-Pb-Au mine is located near Hawk Inlet (58° 05′ 03″ N, 134° 38′ 05″ W) on Admiralty Island in southeastern Alaska, USA. This underground mine produces sulfide- and carbonate-rich, fine-grained tailings, which are filter-pressed to approximately 12 wt. % moisture. Approximately one-half of this material is returned underground for use as structural backfill, while remaining tailings are dry stacked and roller compacted in a tailings storage facility (Figure 1). Tailings pore-water chemistry is largely controlled by the oxidation of sulfide minerals, including pyrite [FeS_2], sphalerite [(Fe,Zn)S], galena [PbS], tetrahedrite [$(Fe,Zn,Cu,Ag)_{12}Sb_4S_{13}$], arsenopyrite [FeAsS] and chalcopyrite [FeCuS], and the dissolution of dolomite [$CaMg(CO_3)_2$] and calcite [$CaCO_3$] during acid neutralization. Dissolved constituents derived from the mill processing circuit, such as SO_4, S_2O_3, Fe and Zn, also influence pore-water chemistry (Condon and Lear, 2006). The tailings deposit is characterized by circumneutral pH drainage conditions and elevated pore-water concentrations of Fe^{II}, Zn, Mn, Ni, As, Sb and Tl.

Figure 1.Tailings storage facility at the Greens Creek mine in 2008.

The Greens Creek mine is located within a temperate coastal rainforest. Mean annual precipitationat the tailings storage facility was 1380 mm from 1997–2005, and the corresponding average annual air temperature was6.0°C. Near-surface tailings temperatures ranged from -1 to +21°C during the study.The magnitude of the seasonal variation in temperature decreased with increasing depth.A median temperature of +8°C was observed within the upper 400 cm. Unsaturated hydraulic conductivities measured using a tension infiltrometer ranged from from 10^{-8} to 10^{-5} cm s^{-1}within the cells and estimated residence times were 4–5 years.

Materials and Methods

Experimental setup

Field-scale experimental cells were constructed within the upper 4 m of the tailings storage facility during October and November 2004. Five test-cell experiments were initiated by excavating a 3 m by 3 m area of freshly placed mill tailings to a total depth of 4 m. An impermeable polypropylene (PP) liner was installed along the excavation walls to constrain flow and transport in a downward vertical direction (Figure 2). Excavated tailings were amended with varied mixtures of locally available OC materials at 5 or 10 vol. %. These mixtures included peat, spent brewing grain (SBG) and municipal biosolids (MB) (Table 1). Amended tailings were inoculated with sulfate-reducing bacteria (SRB) by adding approximately 25 L of peat collected from the anaerobic zone of a tailings drainage retention pond. The cells were subsequently backfilled with OC amended tailings, which were compacted approximately every 50 cm using the excavator bucket. Construction of a control cell (TC2) followed the same methods; however, excavated tailings were not amended with OC or inoculated with SRB.Each test cell and the control cell were instrumented with suction lysimeters and tensiometers, which were installed at 25 and 50 cm intervals to a depth of 400 cm below the tailings surface (Figure 2). Approximately 80 % of suction lysimeters and 50 % of tensiometers were installed in duplicate.

Table 1.Organic carbon amendments evaluated in field trial experiments.

Test Cell	Composition (vol. %)				Description
	Peat	SBG	MB	Tailings	
TC2	0	0	0	100	Excavated control
TC3	5	0	0	95	5 vol. % organic carbon
TC4	2.5	2.5	0	95	5 vol. % organic carbon
TC5	2.5	0	2.5	95	5 vol. % organic carbon
TC6	2.5	1.25	1.25	95	5 vol. % organic carbon
TC7	5	2.5	2.5	90	10 vol. % organic carbon

Sample collection and analysis

Suction lysimeters were used for pore-water collection from 2004 to 2008 (Figure 3). The lysimeters were purged with nitrogen gas, a vacuum of approximately 70 kPa was applied, pore waters were collected over a period of 12 to 18 hours, and the water was purged. This process was repeated twice and samples were transferred into sterile PP syringes during a third collection period. For 2009 and 2010, pore-water samples were extracted from core samples

using an immiscible fluid displacement technique (Figure 3). These core samples were collected using a direct-push method that provided a depth profile of tailings solids and associated pore water (Benner et al., 1999; Hulshof et al., 2006). Three consecutive 150 cm core samples were collected into 7.6 cm diameter Al tubing. These cores were cut into 20 to 30 cm lengths, capped and sealed, and frozen (-20°C) until analysis.

Measurements of pH and redox potential were performed immediately on unfiltered samples according to previously reported methods (Lindsay et al., 2011). Redox potentials were corrected to the standard hydrogen electrode and are reported as Eh values. Remaining water was passed through 0.45 μm (2004–2008) or 0.2 μm (2009–2010) syringe filters. Alkalinity was determined by titration with H_2SO_4 to the bromocresol green – methyl red endpoint, and dissolved H_2S was determined using the methylene blue spectrophotometric method (2005–2008). Samples for anion, cation, trace element, dissolved organic carbon (DOC) and dissolved methane analyses were collected into polyethylene (PE) bottles or amber glass vials, and preserved according to standard methods (APHA, 2005). Samples for $\delta^{34}S\text{-}SO_4$determination were filtered and stored in PE bottles. Ion chromatography (IC) was used for determination of inorganic anion concentrations. Major cations were determined by inductively coupled plasma-optical emission spectroscopy (ICP-OES), and trace element analysis was performed by inductively coupled plasma-mass spectrometry (ICP-MS). Combustion and infrared (IR) detection was used for DOC analysis. Methane concentrations were measured by gas chromatography (GC). Determination of $\delta^{34}S\text{-}SO_4$ values was performed on precipitated $BaSO_4$ using isotope ratio-mass spectrometry (IR-MS).

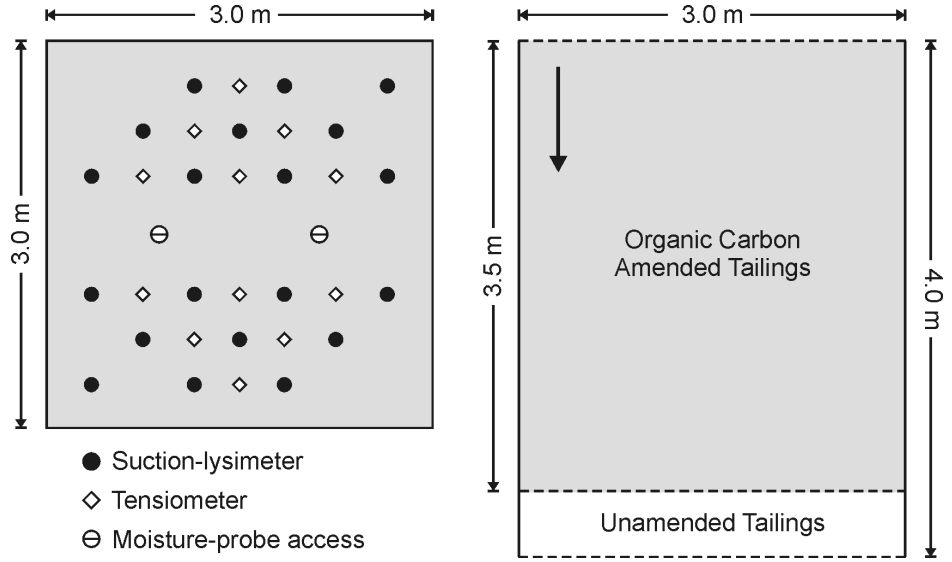

Figure 2. Plan view (left) showing layout of instrumentation, and cross-sectional (right) schematic of experimental cells. Arrow indicates direction of groundwater flow.

Enumerations of SRB, iron-reducing bacteria (IRB) and acid producing (fermentative) bacteria (APB) utilized five-tube most probable number (MPN) techniques. A modified Postgate medium C described by Benner et al. (1999) was used for SRB growth, whereas IRB were grown

in a Fe(III)-EDTA medium developed by Gould et al. (2003). A saccharide- and peptide-rich medium was used to promote growth of fermenters (Hulshof et al., 2003). Inoculation was performed by adding 1.0 g of sample to 9 mL of sterile growth medium. Ten times serial dilutions were performed immediately following inoculation to cover a 10^{10} range in concentration. Samples were incubated at ambient temperature over four weeks for SRB and IRB, and over 96 h for APB. Populations were estimated using an MPN table (Cochran, 1950).

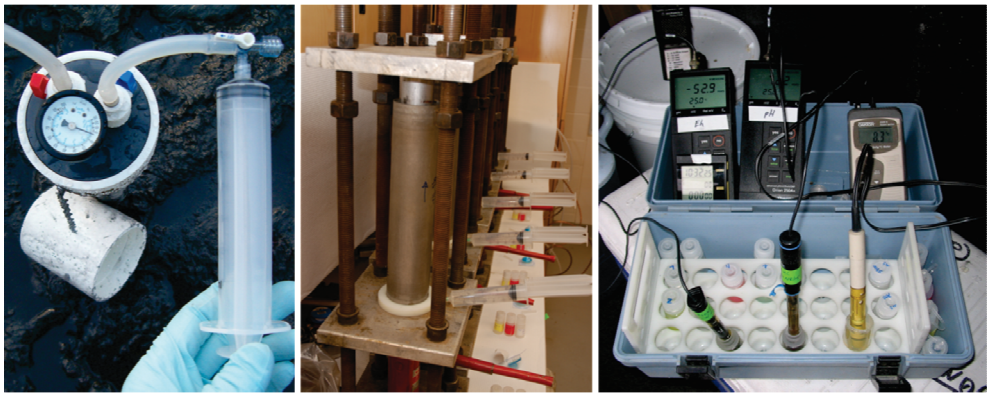

Figure 3.Collection of pore-water from a suction lysimeter (left) and core samples (middle), and setup for measurement of field parameters (right).

Geochemical modeling was performed to assist with interpretation of aqueous geochemical data. The speciation/mass-transfer code MINTEQA2 was used to calculate mineral saturation indices (SIs), which indicate the potential for precipitation or dissolution of a given mineral phase (Allison et al., 1990). The thermodynamic database was modified to be consistent with that of WATEQ4F (Ball and Nordstrom, 1991). Dissolved masses of pore-water constituentsbelow the oxidation zone (i.e., 0.5 to 3.5 m deep) were calculated by trapezoidal integration using measured porosity and moisture content data.

Results and Discussion

Circumneutral pH conditions dominated pore-water chemistry during the experiment. Slight decreases in pH were observed within the upper 1 m of the experimental cells; however, minimum values remained $\geq$ 6.3 throughout the experiment.The lowest pH values within TC2 generally corresponded toelevated pore-water sulfate concentrations and alkalinity, which are indicative of sulfide-mineral oxidation at the tailings surface.Dissolved concentrations of sulfate and metals generally decreased with depth in all cells; however, sulfate removal was apparent in some organic carbon amended cells.

Dissolved organic carbon

Although amendment rates used in these experiments were low, initial increases in DOC concentrations were observed for cells containing SBG and/or MB as a component of the amendment mixture (i.e., TC4–TC7). Average DOC concentrations during Year 1 (2004) ranged from 65 to 370 mg L^{-1} and the highest values were observed in TC7, which contained 10 vol. % OC. Average concentrations in TC4, TC6 and TC7 increased substantially between 2004 and 2005, but subsequently decreased to concentrations lower than those observed in 2004. This decrease was attributed to consumption of labile (i.e., low molecular weight) OC compounds by heterotrophic microbial activity. After 2005, TC4 exhibited relatively consistent average DOC concentrations ranging from 55 to 70 mg L^{-1}. This consistent supply of DOC is indicative of in situ degradation of complex OC compounds (e.g., cellulose) at rates similar to OCconsumption by heterotrophic microogranisms. Variations in DOC concentrations were observed in

TC5– TC7 from 2006–2010, with the highest average concentration observed in 2007. These variations suggest that relative rates of DOC production and consumption varied during the experiment.

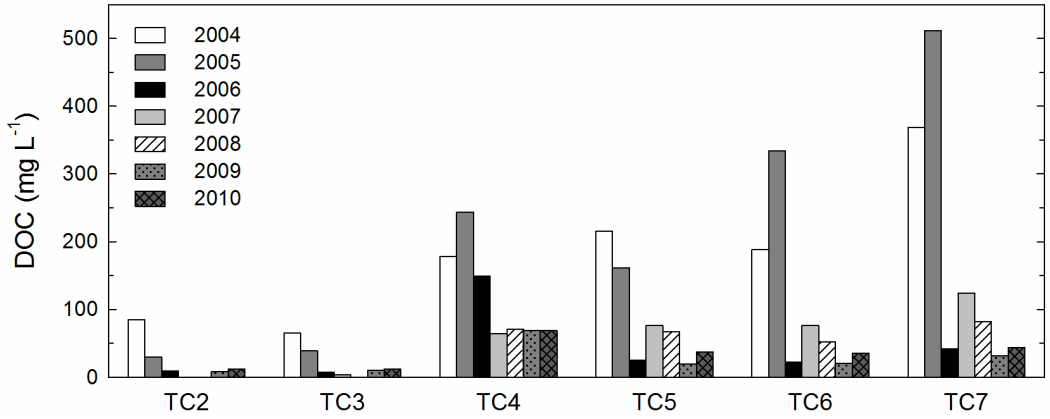

Figure 4.Average concentrations of DOC for test cells from 2004–2010.

Increased DOC concentrations generally corresponded to alkalinity production, suggesting that DOC oxidation was coupled with other biogeochemical processes (Figure 5). This trend was observed throughout the experiment, and the highest DOC concentrations observed in 2010 also corresponded to the highest alkalinity values. This trend was particularly pronounced in TC4, which exhibited the highest average DOC concentration over the final three years of the experiment. Additionally, the lowest 2010 pore-water Ca and sulfate concentrations for TC4 corresponded to the highest DOC concentrations (Figure 5).

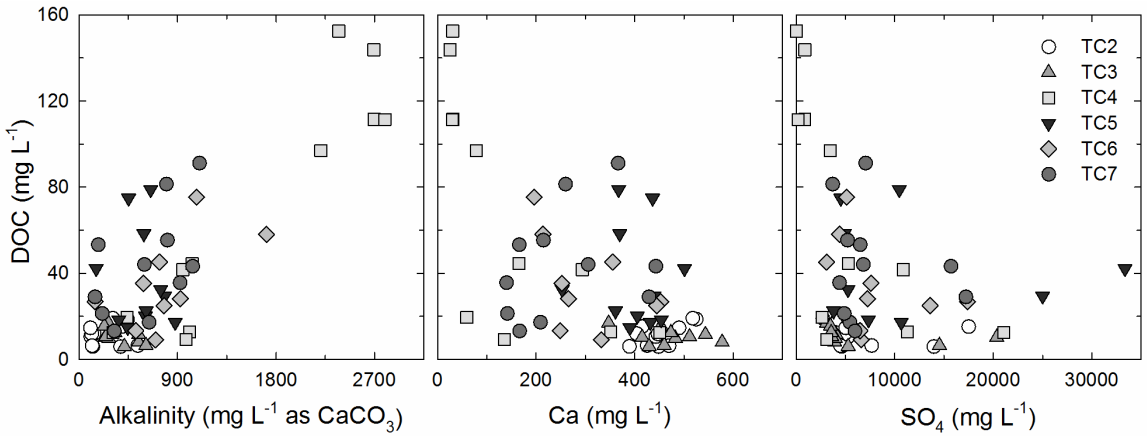

Figure 5. Pore-water DOC concentrations plotted against alkalinity (left), calcium (center) and sulfate (right) for 2010 samples.

Sulfate reduction

Initial sulfate concentrations ranged from 1700 to 3000 mg L^{-1} (average 2100 mg L^{-1}) and were relatively consistent among and within cells (Lindsay et al., 2011). Geochemical modeling indicated that pore water was consistently at saturation with respect to gypsum. Large increases in sulfate concentrations within the upper 0.5 m of each cell with time were attributed to sulfide-

mineral oxidation. Below this depth, elevated pore-water sulfate concentrations persisted through 2005 in all cells, and decreases were observed in TC4, TC6 and TC7 by 2006. Concentrations within these cells decreased to < 500 mg L^{-1} at multiple locations at depths ranging from 2 to 4 m below the tailings surface. Subsequent increases in sulfate concentrations were observed in 2007 for TC6 and 2008 for TC7; however, sulfate removal remained active within TC4 throughout the experiment. After six years (2010), the lowest cumulative dissolved S mass (Table 2) and lowest average SO_4 concentration (Table 3) were observed in TC4. Values of δ^{13}C-DIC (i.e., alkalinity), δ^{13}C-DOC and δ^{34}S-SO_4 measured in 2007 suggest that DSR was coupled with oxidation of OC amendments (Figure 6). Shifts in δ^{13}C-DIC values towards those of the solid phase OC sources (-27.8 to -26.9 ‰) coupled with enrichment of δ^{34}S-SO_4 is indicative of DSR.

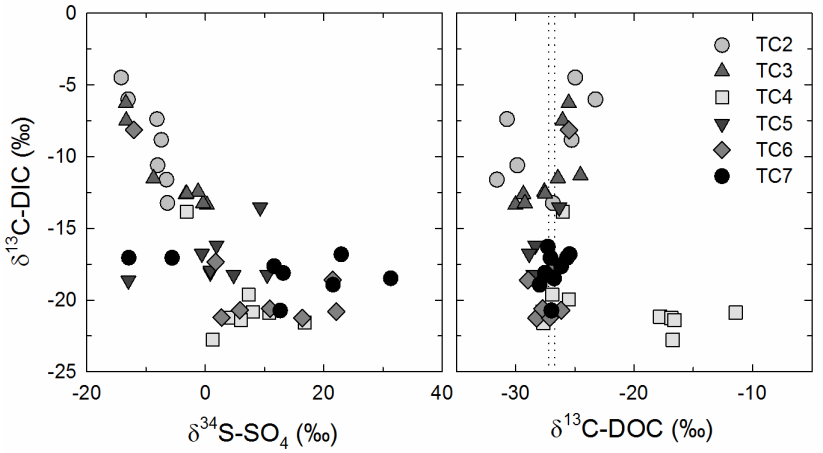

Figure 6. Values of δ^{13}C-DIC plotted against δ^{34}S-SO_4(left) and δ^{13}C-DOC (right) for pore-water samples collected in 2007. Dotted lines indicate range of δ^{13}C values for organic carbon amendments.

Sustained DSR within TC4 was attributed to elevated DOC concentrations and increased SRB activity. A 32 to 62% relative decrease in the total mass of S was observed within TC4 compared to other amended cells and the control (TC2). Declining DOC concentrations and SRB activity within cells that had supported DSR (i.e., TC6, TC7) generally corresponded to increased sulfate concentrations. Furthermore, average sulfate concentrations within thesecells increased above those observed for TC2 and TC3. Modeled SI values suggested that decreases in Ca concentrations were attributed to the precipitation of $CaCO_{3(s)}$rather than gypsumwithin TC5, TC6 and TC7. Removal of Ca via $CaCO_{3(s)}$has potential to limitthe effectiveness of gypsum precipitation as a control on dissolved sulfate concentrations in these cells.

Table 2.Average 2010 pore-water concentrations of sulfate and metals for depths > 40 cm.

Test Cell	Average Concentration (mg L^{-1})						
	SO_4	Fe	Zn	Mn	As	Sb	Tl
TC2	5900	1.1	4.6	0.43	0.017	0.084	0.26
TC3	5000	2.0	9.7	0.60	0.019	0.087	0.075
TC4	3900	1.6	2.2	0.12	0.24	0.33	0.029
TC5	8300	1.6	2.2	0.31	0.086	0.17	0.048
TC6	6400	2.2	2.2	0.37	0.11	0.16	0.013
TC7	6500	0.72	2.7	0.22	0.16	0.33	0.018

Metal removal

Pore-water concentrations of Fe were consistently < 2 mg L^{-1} during initial sample collection in 2004. These low concentrations were attributed to $O_{2(g)}$ introduction and the precipitation of Fe(III) (oxy)hydroxides. Subsequent dissolution of these Fe(III) phases likely resulted inincreased Fe concentrations observed following the addition of OC. The largest increases occurred in cells amended with MB + SBG (i.e., TC6, TC7), and Fe mobilization generally was accompanied by increased populations of iron-reducing bacteria (IRB). Subsequent decreases in Fe concentrations were associated with declining IRB populations and sulfate removal in TC4, TC6 and TC7. However, between 2008 (Lindsay et al., 2009c) and 2010, dissolved Fe concentrations again increased with declining DSR in TC6 and TC7. The lowest cumulative dissolved mass of Fe in 2010 was observed in TC7, whereas this mass was slightly lower for TC4 than TC2 (control) and TC3. Relative increases in dissolved Fe masses in TC5 and TC6 are attributed to the development of iron reducing conditions, coupled with limited DSR and metal-sulfide precipitation over the duration of the experiment.

Table 3. Cumulative dissolved masses of sulfur and metals for 2010 calculated by trapezoidal integration of pore-water chemistry data from 0.4 to 3.5 m below the tailings surface.

Test Cell	Cumulative Dissolved Mass (g)						
	S	Fe	Zn	Mn	As	Sb	Tl
TC2	22000	14	45	4.9	0.17	0.88	2.9
TC3	18000	16	106	7.3	0.17	0.96	0.72
TC4	11000	13	25	2.6	2.8	3.6	0.25
TC5	28000	37	60	6.7	0.81	1.7	0.57
TC6	22000	21	23	3.3	1.09	1.6	0.14
TC7	22000	5.8	30	2.0	1.8	3.1	0.17

Dissolved Zn concentrations initially remained < 11 mg L^{-1}, but increased to > 100 mg L^{-1}in the upper 0.5 m of all cells over time. Decreases in Zn concentrations were observed with depth in all cells. Geochemical modeling indicated that the precipitation of smithsonite [$ZnCO_{3(s)}$] likely contributed to Zn attenuation. However, the precipitation of sphalerite [$ZnS_{(s)}$] was a primary control on Zn concentrations in cells where DSR occurred (Lindsay et al., 2011). The lowest cumulative masses of Zn were associated with TC4 and TC6. Extensive DSR was observed in

these cells, while moderate Fe removal was observed in TC4 in 2010. Relative increases in Zn within TC3 and TC5 compared to the control cell (TC2) are attributed to limited DSR and lower alkalinity values. Similar to Zn, the 2010 cumulative dissolved mass of Mnin TC4 was among the lowest of all cells. Initial Mn concentrations in 2004 were consistently < 3 mg L^{-1}, but increases to > 8 mg L^{-1} were observed by 2005 in cells amended with OC. Subsequent removal of Mn was observed in cells where DSR occurred (i.e., TC4, TC6 and TC7). The precipitation of MnS(s) is not favored at pH < 10 and geochemical modeling results suggest that alkalinity generated via OC oxidation coupled with DSR promoted rhodochrosite [$MnCO_{3(s)}$] precipitation.

The precipitation of Fe(III) (oxy)hydroxide phases following cell construction likely limited initial dissolved As concentrations.Mobilization of As also was observed in test cells amended with OC, which exhibited increased IRB populations and Fe mobilization. The reduction of As(V) to As(III), which exhibits greater mobility in groundwater systems, is thermodynamically favored under iron-reducing conditions (Kocar and Fendorf, 2009). Maximum pore-water As concentrations were observed in 2005 and 2006; however, subsequent decreases from maximum concentrations occurred in cells where DSRoccurred. The precipitation of discrete As-sulfide minerals (e.g., orpiment [As_2S_3] and realgar [As_4S_4]) is favoured under acidic conditions, rather than circumneutral pH conditionsthat were characteristic of pore waters in the test cells. Sorption and co-precipitation reactions with secondary sulfides, such as mackinawite [$Fe_{1+x}S_{(s)}$], pyrite and sphalerite, may contribute to As removal under DSR conditions (Labrenz et al., 2000; Farquhar et al., 2002). AlthoughAs removal was observed in TC4 and TC7 following an initial period of mobilization, these cells exhibited the largest cumulative dissolved masses of As in 2010. Sorption to Fe(III) (oxy)hydroxide likely was an important mechanism of As attenuation in TC2, TC3 and TC5, which exhibited lower cumulative dissolved masses of As in 2010. These phases are subject to reductive dissolution in the presence of OC, presenting a potential mechanism of As mobilization in the future.

Elevated Sb concentrations were observed at the onset of the experiment and are indicative of less effective sorption of Sb(VI) to Fe(III) (oxy)hydroxides at pH > 7 (Leuz et al., 2006). Speciation of Sb was not performed; however, Sb(III) generally dominates under Fe and SO_4 reducing conditions. Reduction of Sb(VI) has the potential to promote attenuation via sorption of Sb(III) to Fe(III) (oxy)hydroxides, which occurs over a broad range of pH (3 to 12). This mechanism may have contributed to Sb removal in cells where limited Fe mobilization occurred. However, (co)precipitation or sorption reactions with secondary Fe-sulfide phases may contribute to Sb removal under DSR conditions (Chen et al., 2003). As of 2010, the greatest cumulative dissolved masses of Sb were observed in TC4 and TC7. These cells previously exhibited similar dissolved concentrations to those of the control cell; however, relative increases over time indicate that initial Sb attenuation mechanisms were ineffective over longer time periods.

Thallium concentrations decreased substantially in test cells containing OC amendments.Average Tl concentrations decreased by 60 to 90 % under DSR conditions in TC4, TC6 and TC7. Removal of Tl was less effective in TC3 and TC5, whereas, the control cell (TC2) exhibited relatively consistent average Tl concentrations with time. Geochemical modeling results indicated that pore-water was generally undersaturated with respect to $Tl_2S_{(s)}$. Removal of Tl under DSR conditions may be attributed to co-precipitation or sorption reactions with secondary sulfide phases (Laforte et al., 2005). Consistently elevated Tl concentrations in TC2–TC3 and TC5 indicate that sorption of Tl to OC and Fe(III) (oxy)hydroxides did not effectively limit Tl mobility. Cumulative pore-water masses of Tl in cells where DSR was observed were approximately ten times lower than the control cell in 2010, six years after initiating the experiment.

Conclusions

Results of this seven-year field study demonstrate that amendment of tailings with a small dispersed mass of organic carbon can support sulfate reduction and pore-water treatment. Amendment of freshly deposited tailings supported increased populations of sulfate-reducing bacteria, dissimilatory sulfate reduction, the precipitation of secondary metal-sulfide minerals and increases in carbonate alkalinity. Consequently, decreases in pore-water concentrations of sulfate, iron, zinc, manganese and thallium were observed under sulfate-reducing conditions. The addition of spent brewing grain and/or municipal biosolids also supported dissimilatory iron reduction resulting in mobilization of iron and arsenic; however, subsequent decreases from maximum concentrations were associated with dissimilatory sulfate reduction.

The influence of sulfate reduction on tailings pore-water chemistry varied among the organic carbon amendments evaluated in this study. Over the duration of this study, conifer-derived peat was generally ineffective for inducing and supporting dissimilatory sulfate reduction. This source of organic carbon, which was evaluated in each of the field trail cells, is comprised of complex ligno-cellulose carbon molecules. The refractory character of peat was evident in the general absence of sulfate reduction within tailings amended with this organic carbon source.In contrast, spent brewing grain was a common component of organic carbon amendments that supported extensive sulfate reduction, metal-sulfide precipitation and alkalinity generation. Production of labile organic compounds (e.g., acetate) by fermentative bacteria likely sustained growth of sulfate-reducing bacteria in cells amended with this organic carbon source. Although reducing conditions developed rapidly in test cells amended with municipal biosolids, sulfate reduction was short lived.Sulfate removal was sustained for a longer time period when spent brewing grain was included in the amendment, yet DSR became limited in TC7 over time.

Amending tailings with organic carbon at a rate of 5 vol. % has potential to support sulfate reduction and overall improvements in the quality of tailings pore water. Organic carbon materials that primarily consist of highly labile (i.e. municipal biosolids) and refractory (i.e. conifer-derived peat) organic carbon are likely to be ineffective as individual amendments. The reactivity of the organic carbon sources may vary temporally and/or spatially, and such variations will likely influence the potential for sulfate reduction and therefore tailings pore-water treatment. Tailings mineralogy will vary according to the composition of the orebody. Therefore, achievable improvements in pore-water quality may vary, both spatially and temporally, within a tailings facility. Amending weathered tailings with organic carbon may result in increases in the mobility of some elements. Accumulated iron(III) (oxy)hydroxide minerals are subject to reductive dissolution in the presence of organic carbon. Liberation of iron, arsenic and other metal(loid)s associated with these secondary solids may have a negative impact on tailings pore-water quality.Despite these potential limitations, this approach presents a potential technique for managing pore-water and drainage quality in mine tailings deposits.

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