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Impacted by Mine Drainage

By:

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Spatial Variations in Water Composition at a Lake Impacted by Mine Drainage

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Abstract

Release of acid drainage from mine-waste disposal areas is a problem of international scale. Contaminated surface water derived from mine wastes originates as direct surface runoff, and indirectly, as subsurface groundwater flow. At Camp Lake, a small Canadian Shield lake in northern Manitoba, direct and indirect release of drainage from an adjacent sulfide-rich tailings impoundment has severely affected lake water quality. Concentrations of sulfide oxidation products are extremely high in the pore waters of the tailings impoundment (pH <1, up to 129000 mg L⁻¹ Fe, 280000 mg L⁻¹ SO₄, 55000 mg L⁻¹ Zn, 5700 mg L⁻¹ Al, 1620 mg L⁻¹ Cu, 97 mg L⁻¹ Cd, and 55 mg L⁻¹ As). Groundwater and surface water derived from the impoundment discharge into a semi-isolated shallow bay in Camp Lake. The release of this water has altered the composition of the lake water, which has modified the physical limnology of the lake. Geochemical profiles of the water column indicate that, despite its shallow depth (6 m), the bay is stratified throughout the year. The greatest accumulation of dissolved metals and SO₄ is in the lower portion of the water column, with concentrations up to 8500 mg L⁻¹ Fe, 20,000 mg L⁻¹ SO₄, 30 mg L⁻¹ Zn, and 100 mg L⁻¹ Al, including elevated concentrations of Cu, Cd, Pb, and Ni. Meromictic conditions and very high solute concentrations are limited to the bay. Outside the bay, solute concentrations are lower and some stratification of the water column exists. Identification of locations and composition of groundwater discharge relative to lake bathymetry is a fundamental aspect of understanding chemical evolution and physical stability of mine impacted lakes.

1. Introduction

The mining of massive-sulfide ore bodies can result in the exposure of high-sulfide waste in the form of mine tailings, waste rock, and mine workings. The oxidation of sulfide minerals can result in the development of low quality water characterized by high concentrations of Fe(II), SO₄, heavy metals, and other constituents. Infiltration of rain and snow melt eventually displaces this metal-rich water into adjacent surface water and underlying aquifers (Nordstrom and Alpers, 1999; Johnson et al., 2000; Blodau 2004). When groundwater affected by sulfide oxidation discharges to a surface water body, Fe(II) can further oxidize, producing additional acidity, and leading to serious environmental degradation (Dubrovsky et al., 1984).

The impact of mine drainage to surface waters has been well documented for lakes (Herlihy and Mills, 1985; Peine and Peiffer, 1998; Blodau et al., 1998; Bachmann et al., 2001; Frommichen et al., 2003) and pit lakes (Davis and Ashenberg, 1987; Doyle and Runnels, 1997; Levy et al., 1997; Eary 1999; Shevenell et al., 1999; Ramstedt et al., 2003; Knoller et al., 2004; Denimal et al., 2005). The purpose of this study was to investigate the geochemical evolution of a lake which has received mine drainage for over 70 years. Particular objectives were to determine the physical mechanisms and geochemical reactions controlling the spatial distribution of dissolved metals and sulfate within the lake.

The study area is at the former Sherritt Gordon Cu-Zn mine in Sherridon, Manitoba, 800 km northwest of Winnipeg (Fig. 1). Sherridon is in the continental, high-boreal region of Canada. The area has a mean annual lake evaporation of 487 mm (Hydrological Atlas of Canada, 1978) and a mean annual precipitation of 478 mm (Environment Canada, 2005). The

average annual temperature in the area is 0.1 °C, and 28 % of the annual precipitation occurs as snowfall.

The Sherritt Gordon mine operated between 1931 and 1951, during which ~7.4 million tonnes of tailings were generated (Goetz and Froese, 1982). The tailings contain up to 30 wt. % S, mainly as pyrrhotite $[\text{Fe}_{1-x}\text{S}]$ and pyrite $[\text{FeS}_2]$, with pyrrhotite equaling or exceeding pyrite in all samples. Other sulfide minerals in the tailings are sphalerite $[(\text{Zn},\text{Fe})\text{S}]$, chalcopyrite $[\text{CuFeS}_2]$, marcasite $[\text{FeS}_2]$, and trace amounts of arsenopyrite $[\text{FeAsS}]$ and galena $[\text{PbS}]$. The primary non-sulfide gangue assemblage of the tailings reported by Moncur et al. (2005) consists mainly of quartz, K-feldspar, albite, and biotite. Other primary minerals, mostly in accessory amounts (1–2%) are chlorite and amphibole. Trace amounts of talc $[\text{Mg}_3\text{Si}_4\text{O}_{10}(\text{OH})_2]$, ilmenite $[\text{FeTiO}_3]$, magnesite $[\text{MgCO}_3]$, sillimanite $[\text{Al}_2\text{SiO}_5]$, rutile $[\text{TiO}_2]$, titanite $[\text{CaTiSiO}_5]$, gahnite $[\text{ZnAl}_2\text{O}_4]$, and garnet $[\text{Fe}^{2+}_3\text{Al}_2(\text{SiO}_4)_3]$, with a few occurrences of cordierite $[\text{Mg}_2\text{Al}_4\text{Si}_5\text{O}_{18}]$, epidote $[\text{Ca}_2\text{Al}_2(\text{Fe}^{3+},\text{Al})\text{Si}_3\text{O}_{12}(\text{OH})]$, siderite $[\text{FeCO}_3]$, clinopyroxene $[(\text{Ca},\text{Na})(\text{Mg},\text{Fe},\text{Al})(\text{Si},\text{Al})_2\text{O}_6]$, and hematite $[\text{Fe}_2\text{O}_3]$ were observed in the tailings.

Tailings were discharged to two separate impoundments (Camp Tailings and Woods Tailings) covering a total area of 47 ha (Fig. 1). Between 1931 and 1932, discharge into Camp Lake infilled half of Camp Bay and formed a 7 ha mounded beach of exposed tailings and an extensive underwater alluvial fan. Groundwater and surface water discharge from the Camp impoundment directly to Camp Lake (Moncur et al., 2003). The Camp Tailings have oxidized for more than 70 years. The remaining tailings were deposited in the Woods impoundment and have been exposed for more than 50 years. The Woods Tailings encompass an area of 40 ha and almost completely fill Trap Lake and about half of Woods

Lake. The majority of the groundwater and surface water derived from the Woods Tailings area is temporarily collected in Woods Lake, from where it eventually discharges into Camp Bay via Woods Weir. Drainage from the northwestern portion of the Woods Tailings and adjacent waste-rock piles accumulates in the Acid Pond that is separated from Camp Bay by the mine road.

Camp Lake is small and narrow. It is 2.75 km in length and consists of two basins and a semi-isolated bay (Fig. 1). The lake covers 1.2 km², has a mean depth of 4.3 m, a maximum depth of 10.5 m, a total volume of 2.7×10^6 m³, and a relative depth (the maximum depth as a percentage of the mean lake diameter) of 0.86 %. Approximately 85 % of the shoreline consists of gneissic bedrock that is stained rusty red along the high waterline. The remaining 15 % of the shoreline is mine tailings and waste rock. The lake is ice-covered six to seven months annually.

The main surface water input to Camp Lake is stream flow from Sherlett Creek and Woods Creek (Woods Weir) (Fig. 1). Surface water discharges from the north end through Camp Weir. The residence time of water in the lake is 93 days, assuming a complete replacement of the water column.

2. Materials and Methods

2.1 *Physiographic Survey*

A bathymetric survey of Camp Lake was conducted during summer 2001 to determine the morphology of the lake using a Lowrance X-16 depth sounder and a hand-held Eagle Expedition GPS. The software package SURFER (Golden Software, Golden, CO) was used

in conjunction with the bathymetry data to construct a bathymetric map of the lake and to calculate surface areas and volumes.

2.2 Water Quality

2.2.1 Camp Lake

Water samples from the Camp Lake water column were collected on two occasions during 2001 and 2002, and one occasion in 2003 beneath 1 m of ice. During the August 2001 sampling session, water was collected from locations CLS1 to CLS8 (Fig. 1). At each location, in-situ measurements of pH, Eh, conductivity, dissolved oxygen (O_2), and temperature were measured from the surface of the water column to the lake bottom using a Hydrolab model H₂O at 10 cm intervals. Dissolved O_2 was not measured in 2001 because of a malfunctioning probe. Prior to sampling, the Hydrolab was calibrated in the field by using standard buffer solutions at pH 7, 4, and 1, and Zobell's solution (Nordstrom, 1977) for pH and Eh, respectively. Water samples for analysis of anions and cations were collected from each location at 1 m intervals, from the surface of the water column to the bottom, using a 3 L PVC Van Dorin Bottle Sampler. The bottle sampler was rinsed with distilled water between each sampling interval. A sealed flow-through cell, maintained at lake temperature, was attached to the bottle sampler to measure independently the pH and Eh before each sample was removed for comparison with the Hydrolab measurements. The pH was measured using an Orion Ross combination electrode (model 815600) and Eh was measured using an Orion platinum redox electrode (model 96-78BN) calibrated in the same manner as the Hydrolab profiles. Measurements of alkalinity were made on filtered samples using a Hach digital titrator and bromcresol green/methyl red indicator and with 0.16 N H_2SO_4 .

Water samples were collected from the bottle sampler in 60 mL syringes and passed through 0.45 μm cellulose-nitrate filters into prewashed 60 mL Nalgene sample bottles. One sample was acidified with 12 N trace-metal grade HCl to a pH of <1 for cation analysis. The second sample was left unacidified and was used for anion analysis. All samples were immediately refrigerated until analysis. Density measurements were made on the water samples by measuring an accurate volume of sample using a Hamilton digital diluter (± 0.01 mL), measuring the mass on a Sartorius Balance (± 0.001 g), then comparing it to double-deionized water using the same method, all maintained at room temperature.

In June 2002, locations CLS1, CLS2, CLS9, and CLS10 were sampled as described above. Sampling during the April 2003 session was conducted using a peristaltic pump and polyethylene tubing lowered through the ice at 1 m intervals at locations CLS1 and CLS2. Measurements of Eh and pH were made in a sealed flow-through cell. Dissolved O_2 was measured using CHEMets[®] kits, for low ($0\text{--}1.0$ mg L^{-1} O_2) and for high ($1\text{--}12$ mg L^{-1} O_2) concentrations.

A peeper was installed adjacent to location CLS3 in 2001. The peeper extended from 12 cm above the lake bottom to 48 cm below the lake bottom. The peeper was constructed, installed and sampled according to the method of Pedersen et al., (1993). The peeper was allowed to equilibrate for 14 days before sampling. A core sample was extracted adjacent to the peeper to provide measurements of pH and Eh.

2.2.2 Streams and Acid Pond

Historical geochemical data for Sherlett Creek, Camp Weir, and Woods Weir summarized here were collected during studies in 1982 (Beck, 1982), 1985 (Acres, 1986), 1992 to 2000 (Manitoba Conservation, unpublished data), and during the present study; 2001 to 2003. Most of the historical data for surface-water quality are limited to pH values and concentrations of selected ions. Water samples were collected from the Acid Pond in August 2000, July 2001, and June 2002.

2.3 Analytical Methods

The National Laboratory for Environmental Testing, Burlington, Ontario, determined Al, As, Ba, Be, B, Cd, Ca, Cr, Co, Cu, Fe, Pb, Mg, Mn, Mo, Ni, K, Si, Na, Sr, V, and Zn in water samples by inductive coupled plasma optical emission spectrometry (ICP-OES). Concentrations of SO_4 were determined using an automatic colorimetric procedure or ion chromatography. Quality control and accuracy evaluation involved analyzing several standards covering a range of concentrations, analysis of field replicates and splits, and addition of standards to unknown samples to assess for matrix effects.

2.4 Geochemical Modeling

Water chemistry was interpreted with the assistance of the equilibrium chemical-speciation/mass-transfer model MINTEQA2 (Allison et al., 1990). The MINTEQA2 data base was modified to make it consistent with that of WATEQ4F (Ball and Nordstrom, 1991). MINTEQA2 was used to calculate the saturation indices for discrete minerals that may be controlling the concentrations of dissolved species in the Camp Lake water column.

2.5 Lake Solids

Two undisturbed sediment cores from Camp Lake were collected from location CLS2 and one core from the Acid Pond in June 2002 using an open-barrel gravity corer with plastic tubing 4.5 cm in diameter. One core from each location was extracted and immediately frozen and shipped to the University of Waterloo for solids analysis. The mineralogy of the lake sediments was studied by optical microscopy in reflected and transmitted light, by scanning electron microscopy (SEM) and energy-dispersion (EDS) analyses, and by qualitative X-ray diffractometry (XRD). Acid volatile sulfides (AVS) within the lake sediments were determined using the technique of Morse and Cornwell (1987). Samples of the Acid Pond and lake solids were also analyzed for total metals by using HF/HNO₃ extraction followed by ICP-OES analysis. The second core extracted from location CLS2 was immediately extruded at 1 cm intervals in the field to measure the pore water pH and Eh.

3. Results and Discussion

3.1 Camp Tailings

A hydrogeochemical investigation of the Camp Tailings (Moncur et al., 2005) illustrated that sulfide oxidation had resulted in the generation of low-pH tailings pore water, which contains high concentrations of metals and sulfate (pH <1, 129000 mg L⁻¹ Fe, 280000 mg L⁻¹ SO₄, 55000 mg L⁻¹ Zn, 7200 mg L⁻¹ Al, 1620 mg L⁻¹ Cu, 230 mg L⁻¹ Mn, 105 mg L⁻¹ Co, 97 mg L⁻¹ Cd, 55 mg L⁻¹ As, 15.3 mg L⁻¹ Ni, 5.9 mg L⁻¹ Pb, and 3.14 mg L⁻¹ Cr). Groundwater and surface water from the tailings impoundment discharge directly into Camp Lake. In groundwater sampled from single-drive-point piezometers installed in the upper meter of the water table, the pH of the pore water varied from 3.6 to 4.2, and the highest concentrations of Fe and SO₄ occurred in piezometer S2 near the center of the tailings impoundment (Fig. 2).

Groundwater velocity across the tailings impoundment ranges between 2 and 3 m a⁻¹ (Moncur et al., 2005). The average density of groundwater from the tailings at piezometer nest S1 was 1.2 g cm⁻³, 12 % higher than the average density of sea water (1.035 g cm⁻³). A peeper was installed into the lake bed adjacent to location CLS3, approximately 10 m from the lakeshore. Acidic, oxidized water, which contains very high concentrations of dissolved metals was observed 30 to 50 cm below ground surface (Fig. 3). This water probably represents migration of acid, high TDS groundwater, derived from the tailings impoundment to a discharge point near the base of the lake.

3.2 Sherlett Creek

The water in Sherlett Creek has an average pH of 7.4, and the concentrations of most metals except Cu, Zn, and Al are below the analytical detection limits (Table 1). The increases in Cu, Zn, and Al concentrations are likely related to the use of waste rock and tailings from the Sherritt Gordon Mine for construction material adjacent to Sherlett Creek. Upstream from the sampling station, roads near and crossing Sherlett Creek are constructed from waste rock that contains visible sulfide minerals. Along the bank of Sherlett Creek are small waste-rock and tailings piles. Some of the waste rock and all of the tailings along Sherlett Creek show extensive sulfide oxidation. Concentrations of metals and SO₄ remain below drinking-water standards (Health Canada, 2003) but above aquatic standards for Al, Zn, Cu, and Pb.

3.3 Woods Weir

Overland flow of groundwater and surface water from Woods Tailings discharges into Woods Lake with extremely elevated concentrations of metals and SO₄. Woods Lake drains through Woods Weir directly into Camp Lake (Fig. 1). Water at Woods Weir had a deep red-

wine color, an average pH of 2.6, and high concentrations of dissolved metals (e.g., Fe up to 2480 mg L⁻¹, SO₄ up to 7500 mg L⁻¹, Table 1). The streambed of Woods Weir is coated with a thick accumulation of orange precipitates, characteristic of Fe oxyhydroxides (McKnight et al., 1988).

3.4 Acid Pond

The Acid Pond adjacent to Camp Lake Bay has a deep red color, an average pH of 2.45, an Eh of 600 mV, and contains elevated concentrations of dissolved metals and SO₄ (Table 1). Drainage from waste-rock piles, abandoned mine workings, and the northwestern portion of Woods Tailings discharges to the Acid Pond (Fig. 1). Although the pond has a depth of only 0.75 m, physical and chemical profiling during June of 2002 revealed a stratified water column. Electrical conductivity and density measurements of the pond surface water were 8.5 mS cm⁻¹ and 1.01 g cm⁻³, increasing to 18 mS cm⁻¹ and 1.03 g cm⁻³ at the bottom of the water column. In most cases, concentrations of metals and SO₄ increased by an order of magnitude with depth, but there is little variation in measured pH and Eh values.

The bottom sediments in the Acid Pond are rich in yellow Fe-oxyhydroxide precipitates that are relatively dense and >3 cm in thickness. The Fe oxyhydroxide precipitates contain mostly Fe, along with Al, Zn, Cu, and Pb (Table 2). Geochemical modeling indicates that the Acid Pond water is supersaturated with respect to Fe oxyhydroxides and jarosite, but is undersaturated with respect to ferrihydrite [nominally 5Fe₂O₃•9H₂O]. The water approaches saturation with respect to anglesite [PbSO₄] and gypsum, suggesting that these minerals, along with the Fe oxyhydroxides, could be

precipitating or co-precipitating from solution and controlling concentrations of dissolved Fe, Al, Pb, Ca, and SO₄.

3.5 Camp Weir

Water flowing over the Camp Weir during the study had a consistent chemistry, with a pH of 3.3 and containing dissolved Fe, SO₄, Al, Zn, Cu, and Pb above drinking-water standards (Health Canada, 2003) and well above those of aquatic standards (Table 1). Extensive Fe staining was observed in the vicinity of the weir.

3.6 Camp Bay Water Column

3.6.1 Physical Limnology

Profiles of pH, Eh, and the concentrations of major and minor ions at Camp Bay during the summer (August, 2001 and June 2002) and in the early spring when the lake was still under 1 m of ice (April 2003, referred to as winter sampling) indicate that this part of the basin has a permanently stratified water column (Fig. 4). Lakes that do not completely mix and that maintain a stagnant bottom layer are termed meromictic. The upper well-mixed layer of a meromictic lake, the mixolimnion, is typically more dilute and displays seasonal changes in composition, whereas the bottom layer remains stagnant and is referred to as the monimolimnion (Wetzel, 1983). Between these two layers is the chemocline, a zone in which the concentrations of most dissolved metals increase rapidly with depth.

Profiles of density and conductivity from samples collected during August 2001, June 2002, and April 2003 at CLS2 show that the water column remains stratified throughout the year and clearly define the location of the mixolimnion, chemocline, and monimolimnion

(Fig. 5). The density of the upper 2 m of the water column is 1.001 g cm^{-3} , it increases abruptly to 1.017 g cm^{-3} between 2 and 3 m, and then stabilizes to an average density of 1.023 g cm^{-3} in the lower 3 to 6 m of the lake. The density in the lower 4 m of the water column is far greater than the maximum density of fresh water, 1.000 g cm^{-3} at 3.94°C (Vallentyne, 1957), suggesting that normal seasonal overturn of the water column would not occur below a 2 m depth. Measurements of specific conductivity show a similar trend, abruptly increasing between a depth of 2 and 3 m.

During summer sampling, a thermocline developed in the water column and coincided with an abrupt depletion of dissolved O_2 in the chemocline (Fig. 4). During ice-free conditions, dissolved oxygen concentrations in the mixed layer are maintained near saturation. During the winter, concentrations of dissolved O_2 in the mixolimnion were 60 % lower than in the summer, and a thermocline was not present. Ice cover during the winter isolates the surface water from the atmosphere, resulting in lower concentrations of dissolved O_2 and a constant temperature.

The mean pH in the mixolimnion was 3.4 during the summer, and increased to 6.5 near the water-ice boundary during the winter. The addition of fresh water to the lake during the early stages of snowmelt likely increased the pH in the mixolimnion in the winter, increasing the stability of the stratification (Anderson, 1958). Within the chemocline, the pH decreased with depth to a minimum of 2.90, 2.71, and 2.60 in 2001, 2002, and 2003, respectively. Below the chemocline, the pH increased with depth into the monimolimnion and reached a maximum of 5.95. Alkalinity is depleted in the upper 4 m of the water column. At a depth of 5 m, the alkalinity was 32 mg L^{-1} as CaCO_3 , and increased with increasing

depth to a maximum of 179 mg L^{-1} near the water-sediment interface, which corresponds to the pH increase.

The measured Eh profiles were consistent throughout the study except during the winter, when the Eh in the mixolimnion was 330 mV lower than summer values. Eh measurements indicate oxidizing conditions at the surface of the water column and reduced conditions with depth, ranging between 660 mV and 180 mV. Redox conditions in the mixolimnion and chemocline are likely dominated by the availability of dissolved O_2 . Within the monimolimnion, the redox control is probably governed by dissolved Fe, as was also observed in meromictic lakes by Balistrieri et al. (1994) and Ramstedt et al. (2003).

The concentration profiles of dissolved major and trace elements in Camp Bay resemble the specific conductance profiles, showing an abrupt increase across the chemocline (Fig. 4). The Camp Bay waters are dominated by SO_4 , Fe, Mg, Ca, Al, and Zn (in order of decreasing concentrations), similar to the high concentrations at depth in meromictic pit lakes affected by AMD (e.g., Berkeley pit (Davis and Ashenberg, 1989; Liberty Pit, Miller et al., 1996; and the Spenceville Copper Pit, Levy et al., 1997)). The existence of meromictic conditions in Camp Bay is unusual considering its low relative depth (the maximum depth as a percentage of the mean lake diameter (Wetzel, 1983)). Meromictic lakes, whether natural or artificial (pit lakes), typically have high relative depths (e.g., natural lakes >5 % and meromictic pit lakes 10–40 % (Doyle and Runnells, 1997)), indicating a small ratio of surface area to depth, which helps to maintain the stratified conditions. Most natural lakes with low relative depths of <2 % undergo seasonal mixing of the water column. Camp Bay maintains meromictic conditions despite having a low relative depth of 1.5 % across the bay (Fig. 6). Stratification in the bay is preserved by the high density of the bottom waters. The

potential for breakdown of stratification is governed by the energy imparted to surface waters, and the density difference between the two layers. In turn, the energy imparted to the surface layer is essentially dictated by the surface wave amplitude, which is governed by the wind vector, fetch and duration of wind events. Camp Bay is a fetch-limited system in that the distance across the bay limits the wave amplitude.

The bathymetry and location of Camp Bay likely have contributed to the accumulation and maintenance of very high dissolved-element concentrations at depth. Camp Bay is a semi-isolated basin separated from the main lake by a higher elevation ridge (Fig. 1). Photographs show that, during mining, lake levels were much lower and resulted in Camp Bay being completely cut off from the main basin. A 1932 photo showing the location of mine buildings relative to the shoreline indicates that the lake levels were at least 2 m lower than the current level. In 2001, the maximum water depth measured over the ridge separating Camp Bay from the main lake was 1.5 m, suggesting that only minor decreases in water level would be necessary to isolate the bay. Air photos taken in 1952 show Camp Bay as a separate water body. During these low-water periods, the bay would not be mixing with the main lake basin, resulting in increased concentrations of dissolved metals and SO_4 induced by evapo-concentration.

The semi-isolated nature likely contributed to the increased concentrations in Camp Bay when it was periodically cut off from the main basin, but the high-density water at depth probably is primarily due to the direct discharge of metal-laden surface and high-density groundwater from the abandoned tailings. A distinct, continuous seepage to Camp Bay was observed in the bay about 1 m from shore, along the mine road that separates the Acid Pond from Camp Lake. The seepage originates from the Acid Pond and is fluorescent green in

color. The seepage seems to be density-driven, with flow visibly extending from the seep zone along the lake bottom out to the depths of Camp Bay. Analyses were not conducted on the seepage because of difficulties in collecting samples, but the green color is likely attributable either to elevated concentrations of ferrous sulfate (Davis and Ashenberg, 1989) or Cu. When this high-density water flows from the Camp Tailings discharges to Camp Bay, it may not mix completely with lake water because of the difference in densities. The tailings-derived water likely accumulates at depth and stratifies, particularly when the lake is iced-covered during winter.

3.6.2 Secondary Precipitates

A core extracted from the lake sediments below the CLS2 profile in Camp Bay shows three stratigraphic units. The upper 6 cm of the core is an accumulation of secondary Fe oxyhydroxides. Concentrations of total metals from the lake-bottom sediment indicate that it is composed mainly of Fe, with elevated concentrations of other metals (Table 2). The Fe concentrations in the lake precipitates are visually similar to those of the precipitates in the Acid Pond, but the metal concentrations in the lake precipitates are typically an order of magnitude higher. The differences in metal concentrations between the Acid Pond and Camp Bay precipitates are a function of the pH and redox conditions. The bottom waters in the Acid Pond had a pH of 2.45 and Eh of 600 mV, whereas water collected from the bottom of Camp Bay was mildly reduced, with a pH of 5.95 and an Eh of 180 mV. The low-pH water of the Acid Pond increases the solubility of metal oxyhydroxides (Theobald et. al., 1963), which prevents dissolved metals from precipitating or co-precipitating with oxyhydroxides. The precipitation of Fe oxyhydroxides at a pH above 4 leads to the co-precipitation or adsorption of metals, including Al, Zn, Cu, Pb, Ni, Co, and Cd (Thornber and Wildman,

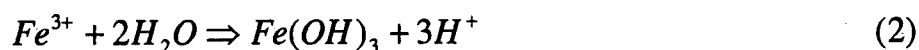
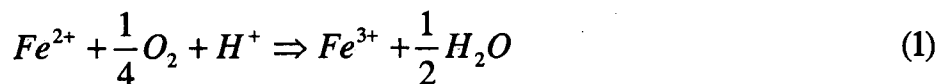
1984; Howell and Bruce, 1995; Holmstrom and Ohlander, 2001). Underlying the Fe-oxyhydroxide precipitates is a 20 cm layer of unoxidized tailings whose mineralogical composition is similar to that of the Camp Tailings (Moncur et al., 2005). Underlying the lake-bottom tailings are native lacustrine detrital sediments.

3.6.3 Fe and SO₄

Profiles of dissolved Fe and SO₄ concentrations strongly resemble those for specific conductivity. The Fe and SO₄ concentrations within the monimolimnion reach maximum values of 8850 mg L⁻¹ and 19300 mg L⁻¹, respectively.

Across redox boundaries, the physical and chemical form of Fe differs. Under oxic conditions, Fe exists as an insoluble Fe³⁺ oxyhydroxide that is capable of strongly adsorbing trace metals (Davison, 1993). Iron in suboxic zones is in its reduced form and exists as dissolved Fe(II). Gravitational settling of Fe oxyhydroxide through an oxic-anoxic interface can result in reductive dissolution, releasing trace elements associated with the Fe oxyhydroxides (Balistrieri, et. al., 1994).

The top of the chemocline in Camp Bay coincides with an increase in dissolved Fe, an abrupt decrease in dissolved O₂, and the lowest pH value measured in the water column. The decrease in pH and depletion of dissolved O₂ in the chemocline is likely a result of Fe(II) oxidation and its subsequent precipitation:





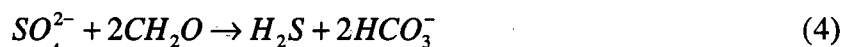
In reaction (1) the oxidation of Fe(II) to Fe(III) consumes dissolved O₂ and 1 mole of acid. The precipitation of Fe oxyhydroxide (2) and Fe hydroxysulfate (3) results in a net release of 3 moles and 6 moles of acid. Although water within the mixolimnion was colorless, it became orange in the active zone of oxidation at the top of the chemocline because of the presence of Fe(III). Directly below the zone of oxidation, the water was green, indicating a mixture of both Fe(II) and Fe(III). Within the monimolimnion, the water was blue-green, suggesting that Fe(II) was dominant.

Saturation indices calculated using MINTEQA2 to determine the minerals that potentially could control Fe and SO₄ concentrations in the Camp Bay water column (Fig. 7) showed that the water is saturated to supersaturated with respect to the Fe minerals ferrihydrite, jarosite, lepidocrocite, and goethite. The shallow water column is undersaturated with respect to siderite, and attains saturation at 5 m depth, consistent with the first occurrence of alkalinity. Saturation with respect to gypsum occurs at 3 m depth, suggesting that gypsum could be controlling SO₄ concentrations.

Mineralogical study of accumulated secondary precipitates within the upper 4 cm of the lake-bottom sediments indicated that the precipitates are Fe oxyhydroxides. The relatively high reflectance of the Fe oxyhydroxides is typical of goethite rather than ferrihydrite, but neither mineral was detectable in a bulk-sample x-ray diffractogram. Four energy-dispersion analyses of the Fe oxyhydroxide consistently detected the presence of S, interpreted to be present as sorbed sulfate on the basis of the homogeneity of the analyzed areas. Up to 3 wt. % of secondary crystals of gypsum were observed within the Fe

precipitate, suggesting that gypsum is one of the principal sinks for sulfate (Fig. 8). No jarosite or carbonate minerals were observed within the Fe precipitates.

Another potential control on the concentration of dissolved Fe and SO₄ in Camp Bay is bacterially mediated sulfate reduction. Sulfate reducing bacteria catalyze the oxidation of organic carbon with the reduction of SO₄ and H₂S (Berner, 1980) through the reaction :



where CH₂O represents a generic organic compound. The release of H₂S in the presence of elevated concentrations of Fe²⁺ can result in the precipitation of sulfide minerals:



where Me²⁺ denotes a divalent metal such as Fe, Cd, Ni, Cu, Co and Zn; and MeS represents a amorphous and poorly crystalline metal sulfide. The sequence of reactions in equations (4) and (5) decreases concentrations of dissolved SO₄, Fe, and other metals, and increases alkalinity and pH (Tuttle et al., 1969; Peine and Peiffer, 1998). The consumption of Fe²⁺ described in reactions (4) and (5) is consistent with the trends of increased pH and alkalinity, and with the decreased Eh and dissolved Fe(II) observed near the bottom of the water column (Fig. 4). Microscopic examination of the lake-bottom Fe precipitates revealed the presence of framboidal pyrite (Fig. 9). Framboidal pyrite is the reaction product of dissolved Fe(II) and reduced sulfur, and forms where sulfate reduction is occurring (Skei, 1987; Pedersen et al., 1993). The presence of framboidal pyrite indicates that sulfate reduction is occurring in the anoxic bottom of Camp Bay. Other studies (e.g. Pedersen et al., 1993; Paktunc et al., 2002) have also reported the presence of framboidal pyrite in sulfate-reducing zones at mining-

impacted sites. Saturation indices calculated using MINTEQA2 indicate that the water column is undersaturated with respect to all sulfide mineral phases in the data base, however AVS was found to be present in the lake bottom precipitates.

Sulfate reducing conditions encountered across the the 6 cm zone immediately below water-sediment interface are also indicated by an abrupt decrease in pore water Eh (Fig. 10). The reduced conditions, presence of framboidal pyrite, appearance of AVS and the decrease in concentration of some metals within the monimolimnion suggests that the lake sediments may be serving as a diffusive sink for some metals. Metal consumption in the sediment pore waters may set up a concentration gradient across the water-sediment interface which drives the diffusion of metals into the sediments. Sulfide precipitation at the water-sediment interface has been proposed to limit the concentration of dissolved metals at many mining impacted systems (Pederson, 1983; Pedersen, et al., 1993; Martin et al., 2001; Martin et al., 2003). The transformation of metals from aqueous phase to solid phase is beneficial in that metals are removed from the water column. These metals, however, may be released back into the water column in the future if geochemical conditions change, for example if a decrease in pH occurs in the monimolimnion.

3.6.4 Al, Zn, and Cu

The concentrations of Al, Zn, and Cu increase at the top of the chemocline, and then abruptly decrease in the monimolimnion. These decreases likely result from the increased pH, which promotes metal removal through precipitation and/or adsorption reactions..

MINTEQA2 calculations indicate that the water approaches equilibrium with respect to gibbsite (Fig. 7) near the bottom of the monimolimnion, consistent with the decline in the

concentrations of dissolved Al. Mineralogical analysis did not detect any secondary Al minerals in the lake-bottom precipitates, but EDS analysis of the Fe oxyhydroxides indicate the presence of sorbed or co-precipitated Al in percent amounts (Fig. 8).

Calculations indicate that the Camp Bay water is undersaturated with respect to all discrete Zn and Cu secondary minerals, including secondary sulfide minerals, and no discrete secondary Zn or Cu precipitates were detected in the mineralogical analysis. Total-metal analysis indicates elevated concentrations of Zn and Cu are present in the lake-bottom precipitates (Table 2). The detection of AVS in the lake sediments suggests that Zn or Cu could be present as secondary sulfides. Another possible sink for dissolved Zn and Cu is through coprecipitation or adsorption reactions within Fe oxyhydroxides limiting dissolved concentrations of the metals (Thornber and Wildman, 1984; Bowell and Bruce, 1995; Holmstrom and Ohlander, 2001).

3.6.5 Trace Metals

Profiles for dissolved concentrations of Pb, Ni, Cr, Cd, and As resemble those for Fe and SO₄. Concentrations of Cr and As were below detection limits during June 2002 and April 2003, possibly because of low dissolved concentrations or analytical uncertainties.

Geochemical model calculations indicate that the water column is undersaturated with respect to discrete Pb, Ni, Cr, Cd, and As minerals. Although Pb, Ni, Cr, Cd, and As were not detected during EDS analysis, Pb, Ni, and Cr are present in bulk-sample chemical analyses of the lake-bottom precipitates (Table 2).

3.7 Camp Lake – Main Basin

Depth profiles of pH, Eh, temperature, and the concentrations of major and minor ions collected in August 2001 at CLS8, the deepest point of the northern basin of Camp Lake, are constant with depth and exhibit no indication of stratification; the results are consistent with a well-mixed water column (Fig. 11). The chemical composition of the water column at CLS8 is similar to those of the Camp Bay mixolimnion and water flowing over Camp Weir during the same time period, suggesting that the shallow surface waters are well-mixed in each of the basins. Water-column profiles along a transect extending from the Camp Tailings outward across Camp Lake (CLS3 to CLS7, Fig. 10) indicate that the water column is well mixed, however, there is some stratification below a depth of 5 m.

The southern basin of Camp Lake has a maximum depth of 10.5 m and was sampled during the summer (August, 2001 and June 2002) and at the end of the winter when the lake was still under 1 m of ice (April 2003). Although solute profiles in the upper 6 m of the water column are similar to those measured at locations CLS3 to CLS8, abrupt increases in pH, dissolved metals, and sulfate, and decreases in Eh, temperature, and concentrations of O₂ occur below 6 m depth (Fig. 13). Comparison of the depth profiles obtained from the different sampling events indicates that the water column remains stratified below 6 m. Results from geochemical modeling of the CLS1 water column also suggest conditions similar to those in CLS2 may be controlling the concentrations of dissolved SO₄, Fe, and other metals (Fig. 14).

Although water from Sherlett Creek discharging into the southern basin is relatively fresh, metals and sulfate originating from Camp Bay seem to have accumulated at the base of the southern basin through density-driven flow between the two basins. It is likely that high-

density water flowing from Camp Bay is accumulating at the bottom of the southern basin in Camp Lake, thereby producing stratification of the water column.

4. Conclusion

Sulfide oxidation reactions occurring within the Sherridon Tailings impoundment have resulted in the release of elevated concentrations of H^+ , SO_4 , and metals. The discharge of groundwater and surface water from the impoundment to Camp Lake has resulted in localized meromictic conditions induced by the dense mine-drainage water. Seasonal changes in the inflow of surface water, groundwater, and wind mixing has no effect on the depth of stratification, which remained constant during the study period despite the lake having a low relative depth, possibly preserved by the high density of bottom waters.

Where meromixis occurred in the lake, the upper well-mixed mixolimnion was more dilute and displays seasonal changes in composition, whereas the monimolimnion remained stagnant with elevated concentration of most metals and SO_4 , except Al, Zn, and Cu which showed a decrease with depth. Between these two layers in the chemocline, concentrations of metals and SO_4 increase abruptly. Secondary minerals that were identified as the primary controls on dissolved metals and SO_4 in Camp Lake were Fe oxyhydroxides and gypsum. The occurrence of secondary framboidal pyrite and acid volatile sulfides in the bottom lake sediments indicates that sulfate reduction is occurring and metals are being removed from the water column as sulfides. Geochemical calculations further suggest that jarosite, and gibbsite may also control metal and SO_4 solubility in the lake.

Acknowledgements

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References

- Acres International Ltd., 1986. Research Program on Abandoned Base Metal Mine Tailings in Manitoba - Final Report, Vol. 1. Unpublished Report. Winnipeg, Manitoba.
- Allison, J.D., Brown, D.S., Novo-Gardac, K.L., 1990. MINTEQA2/PRODEFA2, A Geochemical Assessment Model for Environmental Systems, Version 3.0 User's Manual. Environmental Research Laboratory, Office of Research and Development, U.S. EPA, Athens, GA.
- Anderson, G.C., 1958. Some limnology features of a shallow saline meromictic lake. *Limnol. Oceanogr.* 3, 259-270.
- Bachmann, TM, Friese, K, Zachmann, DW., 2001. Redox and pH conditions in the water column and in the sediments of an acidic mining lake. *J. Geochem. Explor.* 73, 75-86.
- Balistrieri, L.S., Murray, J.W., and Paul, R., 1994. The geochemical cycling of trace elements in a biogenic meromictic lake. *Geochim. Cosmochim. Acta* 58, 3993-4008.
- Ball, J.W., Nordstrom, D.K., 1991. User's manual for WATEQ4F with revised thermodynamic data base and test cases for calculating speciation of major, trace and redox elements in natural waters. U.S. Geol. Surv. Open-File Rep. 91-183.
- Beck, A.E. 1983., Aquatic Impact Assessment of the Sherridon Mine. Water Standards and Studies Report #83-25.
- Berner, R.A. 1980., Early Diagenesis: A Theoretical Approach. Princeton University Press. Princeton, NJ.
- Blodau, C., Hoffmann, S., Peine, A., and Peiffer, S., 1998. Iron and sulfate reduction in the sediments of acidic mine lake 116 (Brandenburg, Germany): Rates and geochemical evaluation. *Water Air Soil Pollut.* 108, 249-270.
- Blodau, C., 2004. Evidence for a hydrologically controlled iron cycle in acidic and iron rich

- sediments. *Aquat. Sci.* 66, 47-59.
- Bowell, R.J., Bruce, I., 1995. Geochemistry of iron ochres and mine waters from Levant Mine, Cornwall. *Appl. Geochem.* 10, 237-250.
- Davison, W. 1993., Iron and manganese in lakes. *Earth-Science Rev.* 34, 119-163.
- Davis, A. and Ashenberg, D., 1987. The aqueous geochemistry of the Berkeley Pit, Butte, Montana, U.S.A. *Appl. Geochem.* 4, 23-36.
- Denimal, S., Bertrand, C., Mudry, J., Paquette, Y., Mochart, M., Steinmann, M. 2005. Evolution of the aqueous geochemistry of the mine pit lakes – Blanzky-Montceau-les-Mines coal basin (Massif Central, France): origin of sulfate contents; effects of stratification on water quality. *Appl. Geochem.* 20, 825-839.
- Doyle, G.A. and Runnels, D.D., 1997. Physical limnology of existing pit lakes. *Mining Eng.* Dec. 76-80.
- Dubrovsky, N.M., Morin, K.A., Cherry, J.A., Smyth, D.J.A., 1984. Uranium tailings acidification and subsurface contamination migration in a sand aquifer. *Water Poll. Res. J. Canada* 19, 55-89.
- Eary, L.E., 1999. Geochemical and equilibrium trends in mine pit lakes. *Appl. Geochem.* 14, 963-987.
- Environment Canada, 2005. Canadian Climate Normals, 1927-1990, Flin Flon, Manitoba. Accessible at: http://www.climate.weatheroffice.ec.gc.ca/climate_normals.
- Frommichen, R., Went-Potthoff, K., Friese, K., Fischer, R., 2004. Microcosm studies for neutralization of hypolimnic acid mine pit lake water (pH 2.6). *Environ. Sci. Technol.* 38, 1877-1887.
- Goetz, P.A., Froese, E., 1982. The Sherritt Gordon massive sulphide deposit. *Geol. Assoc. Can. Special Pap.* 25, 557-569.
- Health Canada, 2003. Federal-Provincial Advisory Committee on Environmental and Occupational Health, Guidelines for Canadian Drinking Water Quality.
- Herlihy, A.T. and Mills, A.L. 1985., Sulfate reduction in freshwater sediments receiving acid mine drainage. *Appl. Environ. Microbiol.* 49, 179-186.
- Holmstrom, H., Ohlander, B., 2001. Layers rich in Fe- and Mn-oxyhydroxides formed at a tailings-pond water interface, a possible trap for trace metals in flooded mine tailings. *J. Geochem. Explor.* 74, 189-203.

- Hydrological Atlas of Canada, 1978. Fisheries and Environment Canada. Surveys and Mapping Branch, Natural Resour. Can., Ottawa.
- Johnson, R.H., Blowes, D.W., Robertson, W.D., Jambor, J.L. 2000. The hydrogeochemistry of the Nickel Rim mine tailings impoundment, Sudbury, Ontario. *J. Contam. Hydrol.* 41, 49-80.
- Knoller, K., Fauville, A., Mayer, B., Strauch, G., Friese, K., Veizer, J., 2004. Sulfur cycling in an acid mining lake and its vicinity Lusatia, Germany. *J. Chem. Geol.* 204, 303-323.
- Levy, D.B., Custis, K.H., Casey, W.H., and Rock, P.A., 1997. The aqueous geochemistry of the abandoned Spenceville Copper Pit, Nevada County, California. *J. Environ. Qual.* 26, 233-243.
- Martin, A.J., McNee, J.J., Pedersen, T.F., 2001. The reactivity of sediments impacted by metal-mining in Lago Junin, Peru. *J. Geochem. Explor.* 74, 175-187.
- Martin, A.J., Jambor, J.L., Pedersen, T.F., Crusius, J., 2003. Post-depositional behavior of Cu in a metal-mining polishing pond (East Lake, Canada). *Environ. Sci. Technol.* 37, 4925-4933.
- McKnight, D.M., and Bencala, K.E., 1988. Diel variations in iron chemistry in an acidic stream in the Colorado Rocky Mountains, U.S.A. *Arct. Alp. Res.* 20, 492-500.
- Miller, G.C., Lyons, W.B., and Davis, A., 1996. Understanding the water quality of pit lakes. *Environ. Sci. Technol.* 30, 118-123.
- Moncur, M.C., Ptacek, C.J., Blowes, D.W., Jambor, J.L., 2003. Fate and transport of metals from an abandoned tailings impoundment after 70 years of sulfide oxidation. In: Graeme, S., Beckett, P., Conroy, H. (Eds.), *Proceedings of Sudbury '03, Mining and the Environment III*. Laurentian Univ., Sudbury, Ontario, pp. 238-247.
- Moncur, M.C., Ptacek, C.J., Blowes, D.W., Jambor, J.L., 2005. Release, transport and attenuation of metals from an old tailings impoundment. *Appl. Geochem.* 20, 639-659.
- Morse, J.W., Cornwell, J.C., 1987. Analysis and distribution of iron sulfide minerals in recent anoxic marine sediments. *Mar. Chem.* 22, 55-69.
- Nordstrom, D.K., 1977. Thermochemical redox equilibria in Zobell's solution. *Geochim. Cosmochim. Acta* 41, 1835-1841.
- Nordstrom, K.D., Alpers, C.N., 1999. Geochemistry of acid mine waters. In: Plumlee, G.S., Logsdon, M.J. (Eds.), *The Environmental Geochemistry of Mineral Deposits. Part A: Processes, Techniques, and Health Issues*. *Rev. Econ. Geol.* 6A, 133-160.

- Paktunc, A.D., and Dave, N.K., 2002. Formation of secondary pyrite and carbonate minerals in the Lower Williams Lake tailings basin, Elliot Lake, Ontario, Canada. *Am. Mineral.* 87, 593-602.
- Pedersen, T.F., 1983. Dissolved heavy metals in a lacustrine mine tailings deposit – Buttle Lake, British Columbia. *Mar. Pollut. Bull.* 14, 249-254.
- Pedersen, T.F., Mueller, B., and McNee, J.J., 1993. The early diagenesis of submerged sulphide-rich mine tailings in Anderson Lake, Manitoba. *Can. J. Earth Sci.* 30, 1099-1109.
- Peine, A. and Peiffer, S., 1998. In-lake neutralization of acid mine lakes. In Geller W., Klapper, H., Salomons, W., (Eds.), *Acid Mining Lakes: Acid Mine Drainage, Limnology, and Reclamation*. Springer. Berlin. pp. 47-63.
- Ramstedt, M., Carlsson, E., and Lovgren, L., 2003. Aqueous geochemistry in the Udden Pit Lake, Northern Sweden. *Appl. Geochem.* 18, 97-108.
- Shevenell, L., Connors, K.A., and Henery, C.D., 1999. Controls on pit lake water quality at sixteen open-pit mines in Nevada. *Appl. Geochem.* 14, 669-687.
- Skei, J.M., 1987. Formation of framboidal iron sulfide in the water of a permanently anoxic fjord – Framvaren, South Norway. *Mar. Chem.* 23, 345-352.
- Theobald, P.K., Lakin, H.W., and Hawkins, D.B., 1963. The precipitation of aluminum, iron, and manganese at the junction of Deer Creek with the Snake River in Summit County, Colorado. *Geochim. Cosmochim. Acta* 27, 121-132.
- Thornber M.R., Wildman, J.E., 1984. Supergene alteration of sulfides, VI. The binding of Cu, Ni, Zn, Co, and Pb with gossan (iron bearing) minerals. *Chem. Geol.* 44, 399-434.
- Tuttle, J.H., Dugan, P.R., and Randles, C.I., 1969. Microbial sulfate reduction and its potential utility as an acid mine water pollution abatement procedure. *Appl. Microbiol.* 17, 297-302.
- Vallentyne, J.R., 1957. Principles of modern limnology. *Am. Sci.* 45, 218-244.
- Wetzel, R.G., 1983. *Limnology*. 2nd edition. Saunders College Publ., Philadelphia.

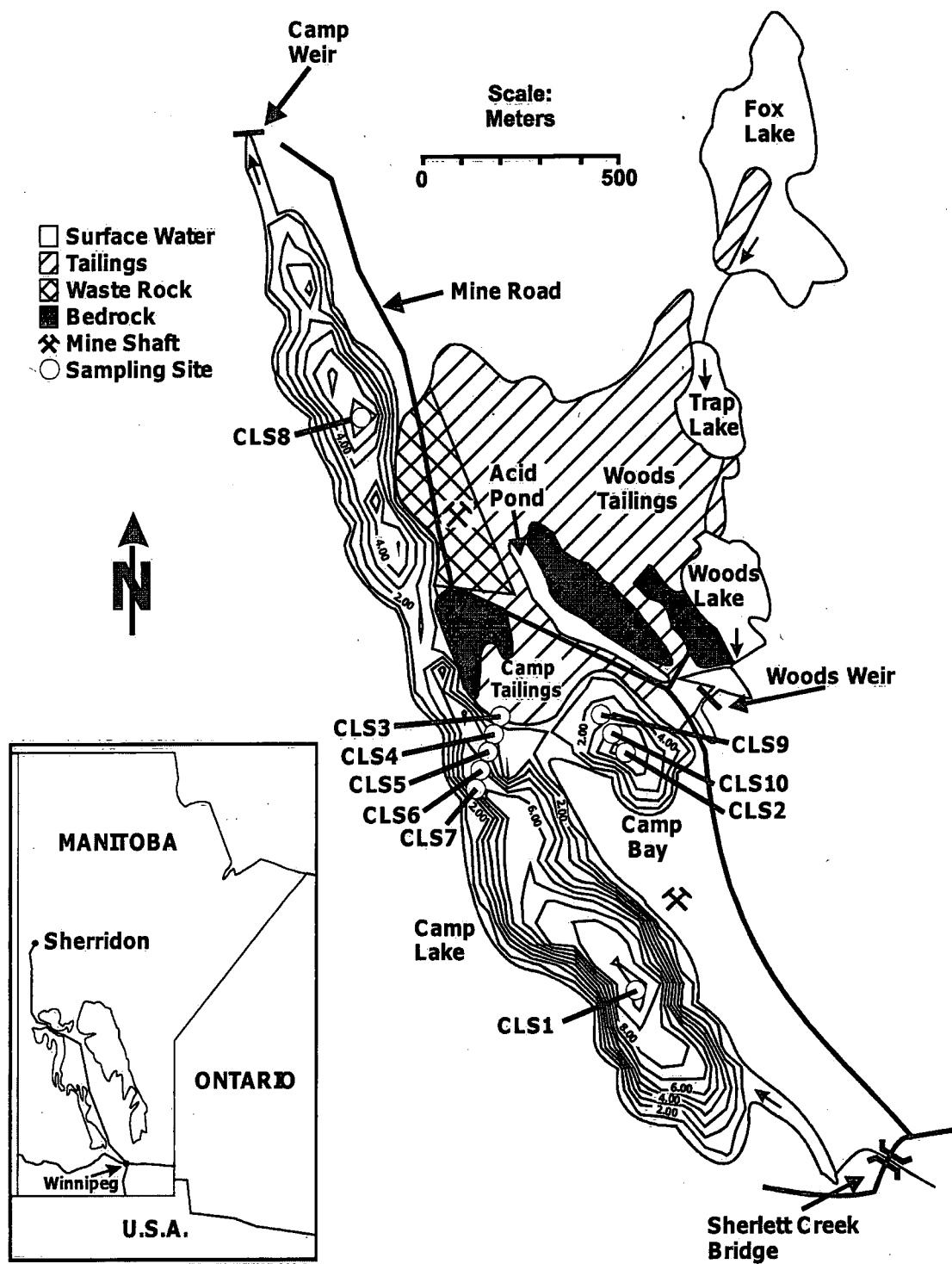


Fig. 1. Map of the study area, showing tailings, lakes, streams, weirs, and mine shafts. Small arrows represent the direction of surface-water flow. Lake contour lines are at 1 m intervals.

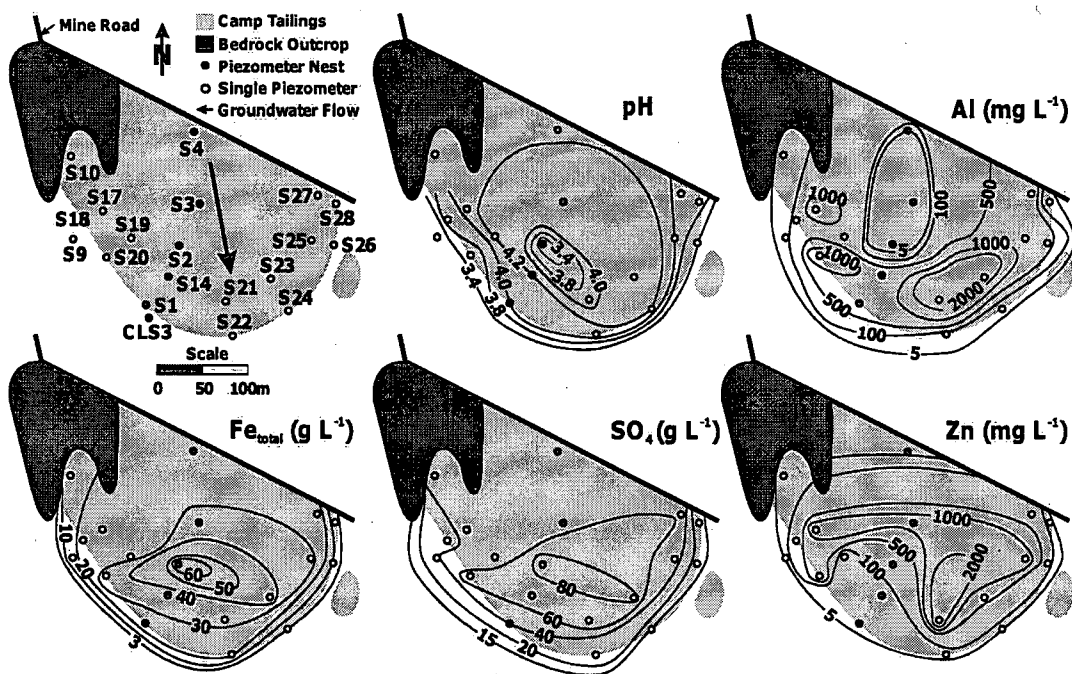


Fig. 2. Location of single-drive-point piezometers and piezometer nests installed in the upper 1 m of the saturated zone of the Camp Lake Tailings. Water chemistry is for samples collected in August 2001. The arrow shows the average direction of groundwater flow as reported by Moncur et al. (2005).

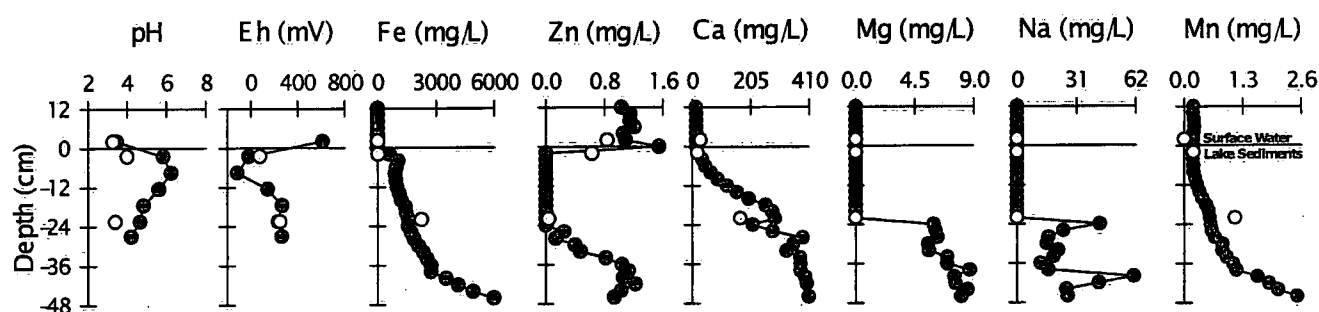


Fig.3 Profile of the lake sediment pore water at sampling location CLS3 below 1 m of surface water. Symbols correspond to: ○ samples extracted from piezometers in 2000; ● samples extracted using a peeper in 2001. The solid line represents the boundary between groundwater and surface water.

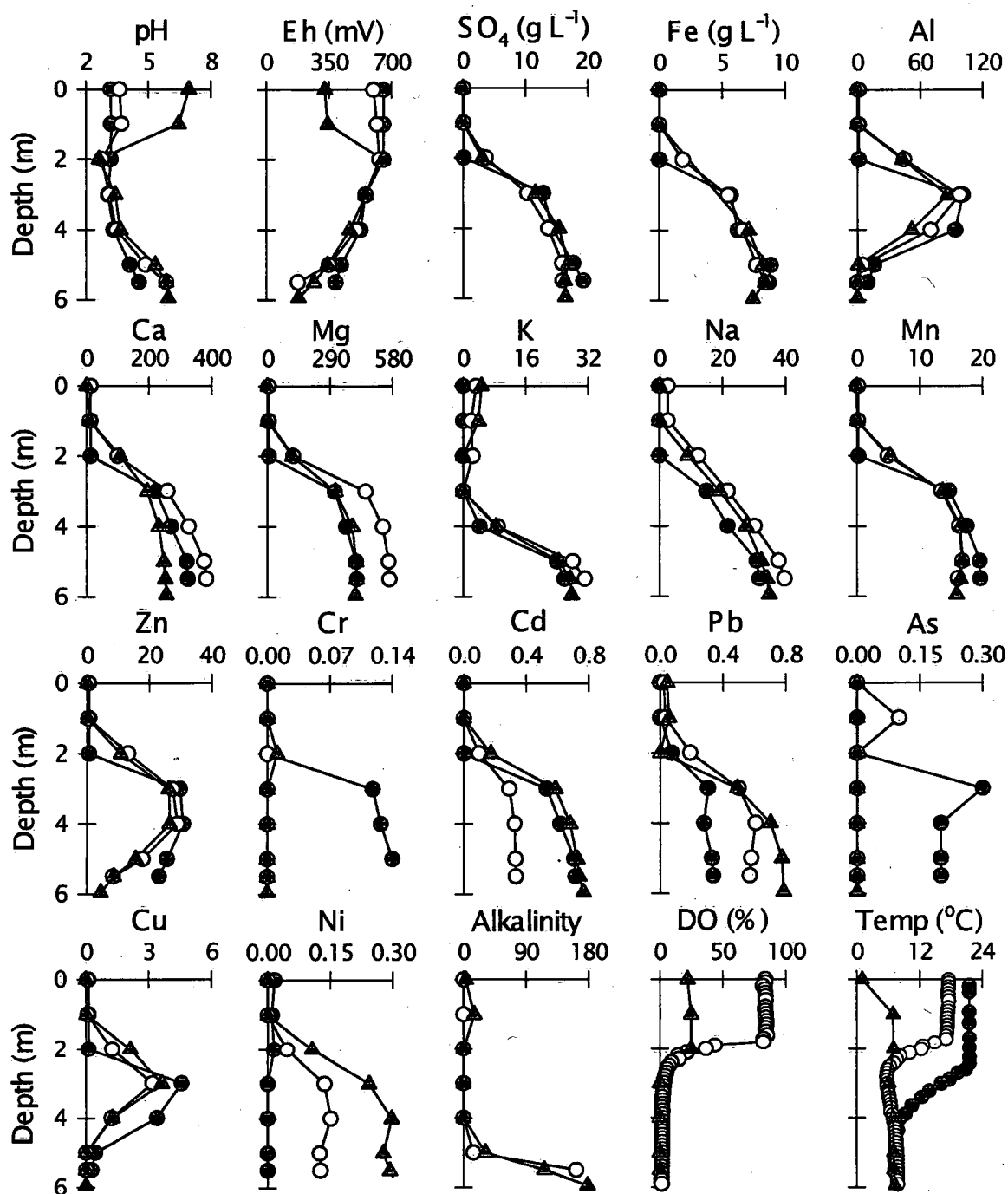


Fig. 4. Depth profiles of water chemistry in the center of Camp Bay (CLS2). Symbols correspond to sampling during: ● August 2001; ○ June 2002; ▲ April 2003 (beneath 1 m of ice). Note that concentrations of SO_4 and Fe are in g L^{-1} and all succeeding values except DO and Temp are mg L^{-1} . DO refers to dissolved oxygen and alkalinity is expressed as CaCO_3 . DO and alkalinity were not measured in 2001.

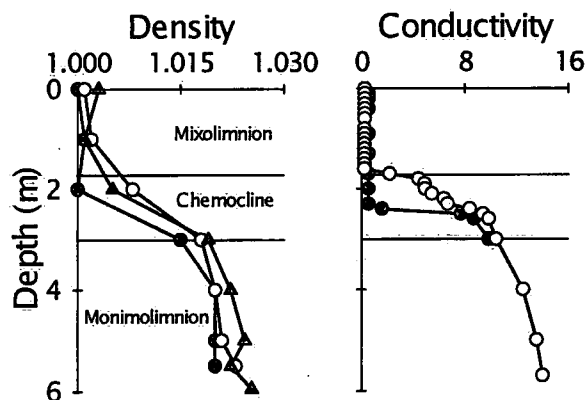


Fig. 5 Depth profiles of density (g cm^{-3}) and specific conductivity (mS cm^{-1}) through Camp Bay (CLS2) delineating locations of the mixolimnion, chemocline and monimolimnion. Symbols correspond to sampling during: ● August 2001; ○ June 2002; ▲ April 2003 (measured beneath 1 m of ice). Conductivity was not measure in 2003.

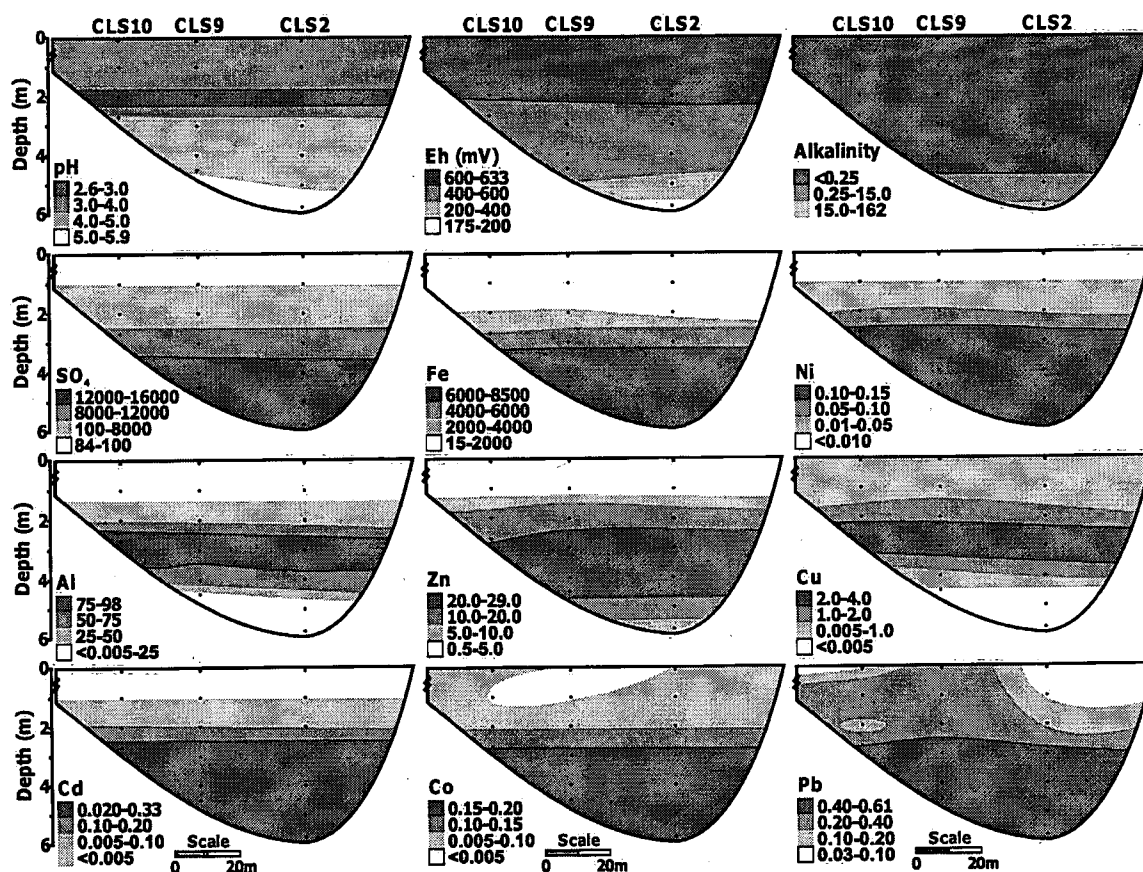


Fig. 6. Water chemistry in a transect across Camp Lake Bay, from CLS10 to CLS2 measured June 2002. Black dots represent sampling locations. Note that concentrations are in mg L^{-1} and alkalinity is expressed as CaCO_3 .

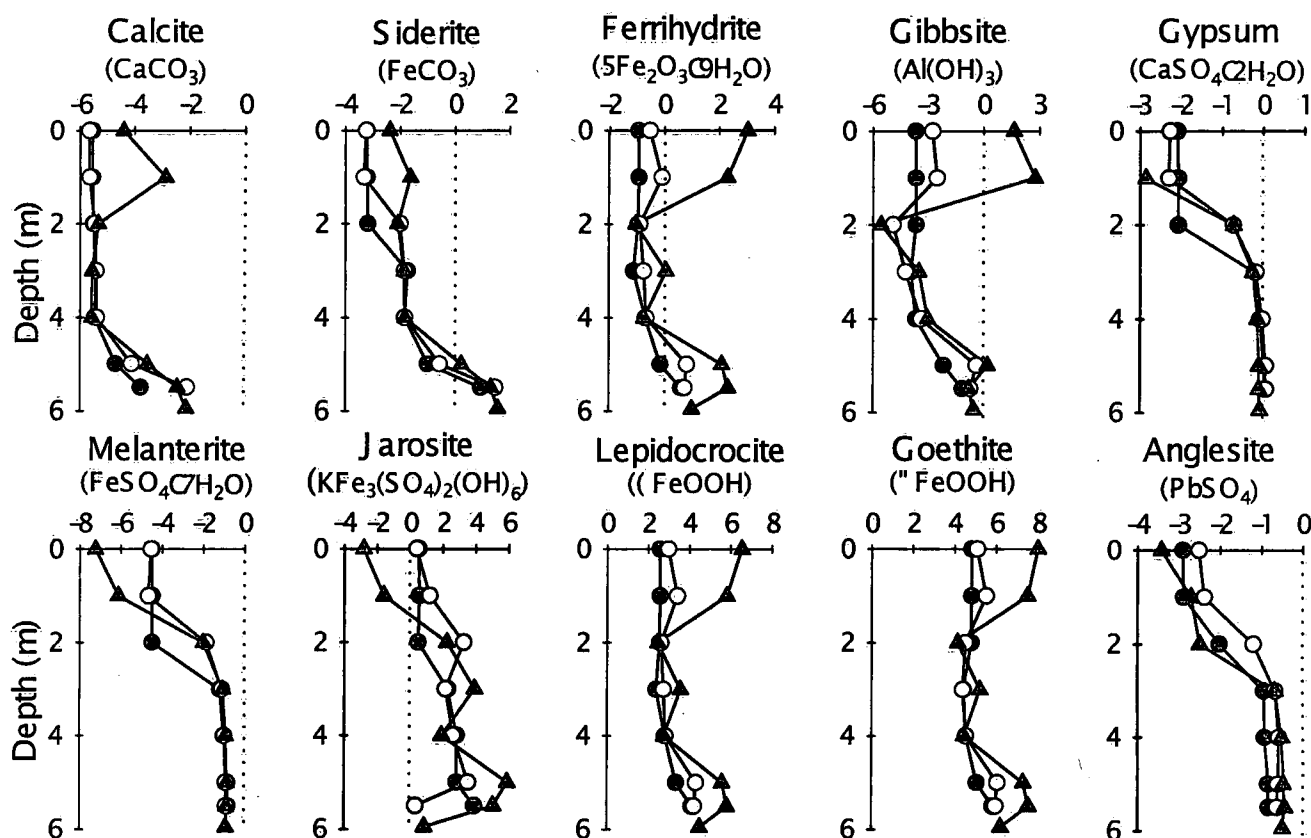


Fig. 7. Saturation indices calculated using MINTEQA2, plotted versus depth at Camp Lake Bay (CLS2). Symbols correspond to sampling during: ● August 2001; ○ June 2002; ▲ April 2003 (beneath 1 m of ice).

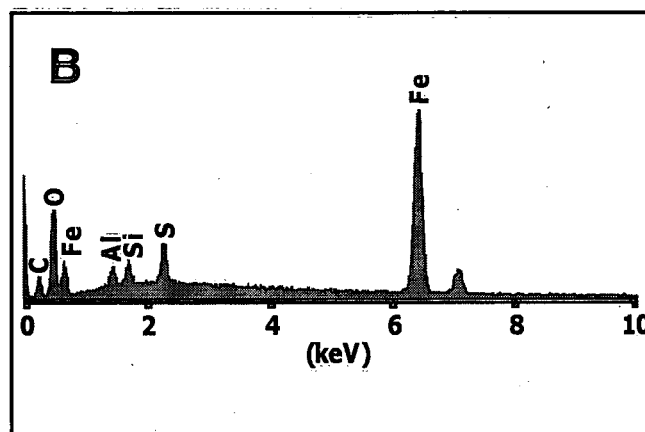


Fig. 8. (A) Photomicrograph in cross-polarized light, internal reflection, showing prismatic secondary crystals of gypsum within a matrix of Fe oxyhydroxide. Width of field 0.625 mm. (B) EDS spectrum for the Fe oxyhydroxide in (A).

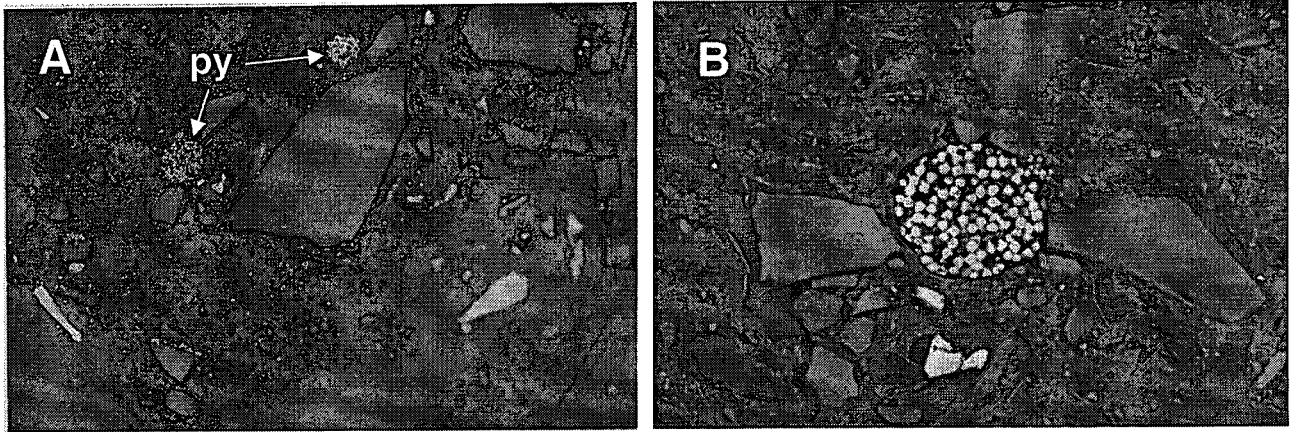


Fig. 9. (A) Photomicrograph in plain reflected light showing framboids of pyrite (py). Width of field 0.3 mm. The BSE image in (B) is a magnification of the framboid at the left arrow in (A) (diameter $\sim 20\ \mu\text{m}$). Samples were extracted from lake-bottom sediments in Camp Bay at location CLS2.

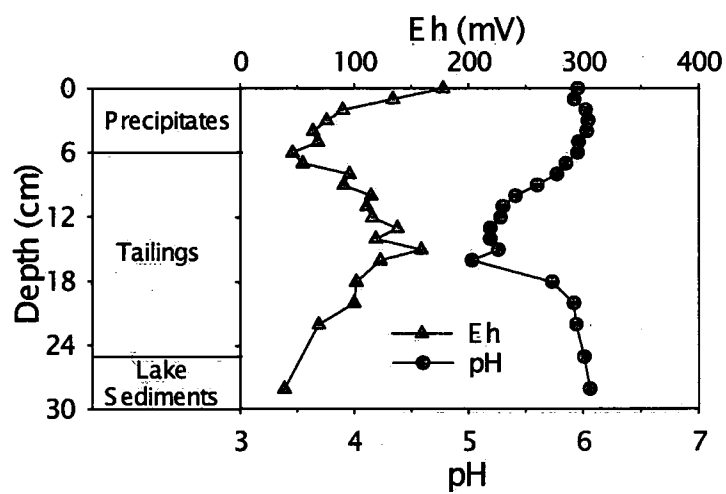


Figure 10. Profile of pore water Eh and pH in the lake sediments at sampling location CLS2.

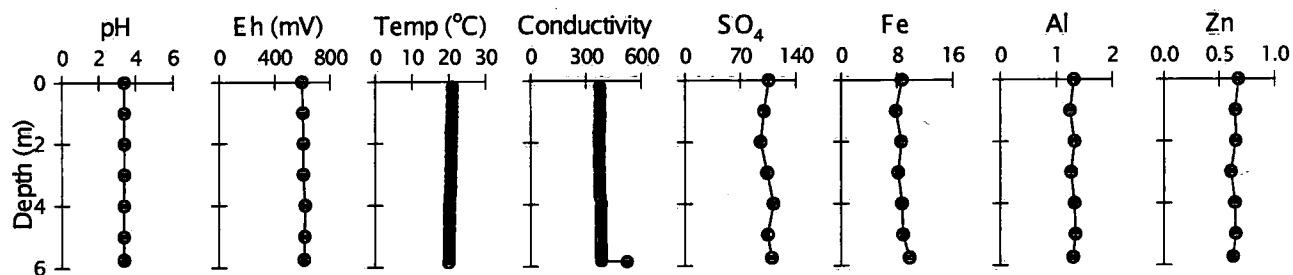


Fig. 11. Depth profiles of select water chemistry through the northern basin (CLS8) in Camp Lake, measured August 2001. Conductivity is in $\mu\text{S cm}^{-1}$, and SO_4 , Fe, Al, and Zn are in mg L^{-1} .

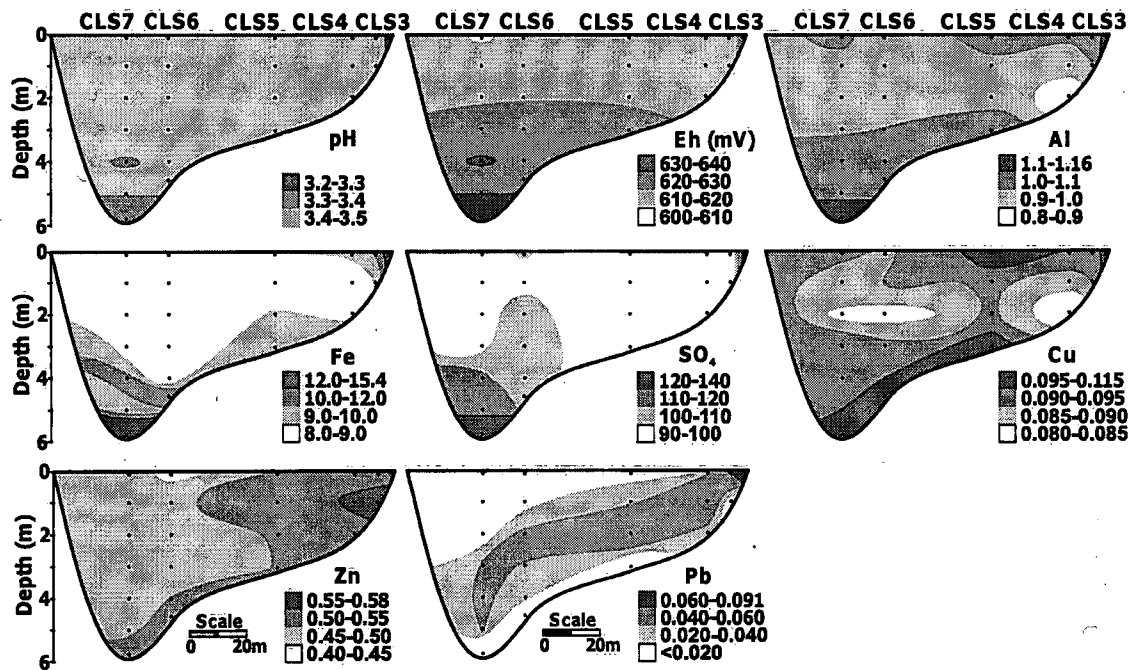


Fig. 12. Water chemistry in a transect across Camp Lake, from CLS3 to CLS7 measured August 2001. Black dots represent sampling locations. Note that concentrations are in mg L⁻¹.

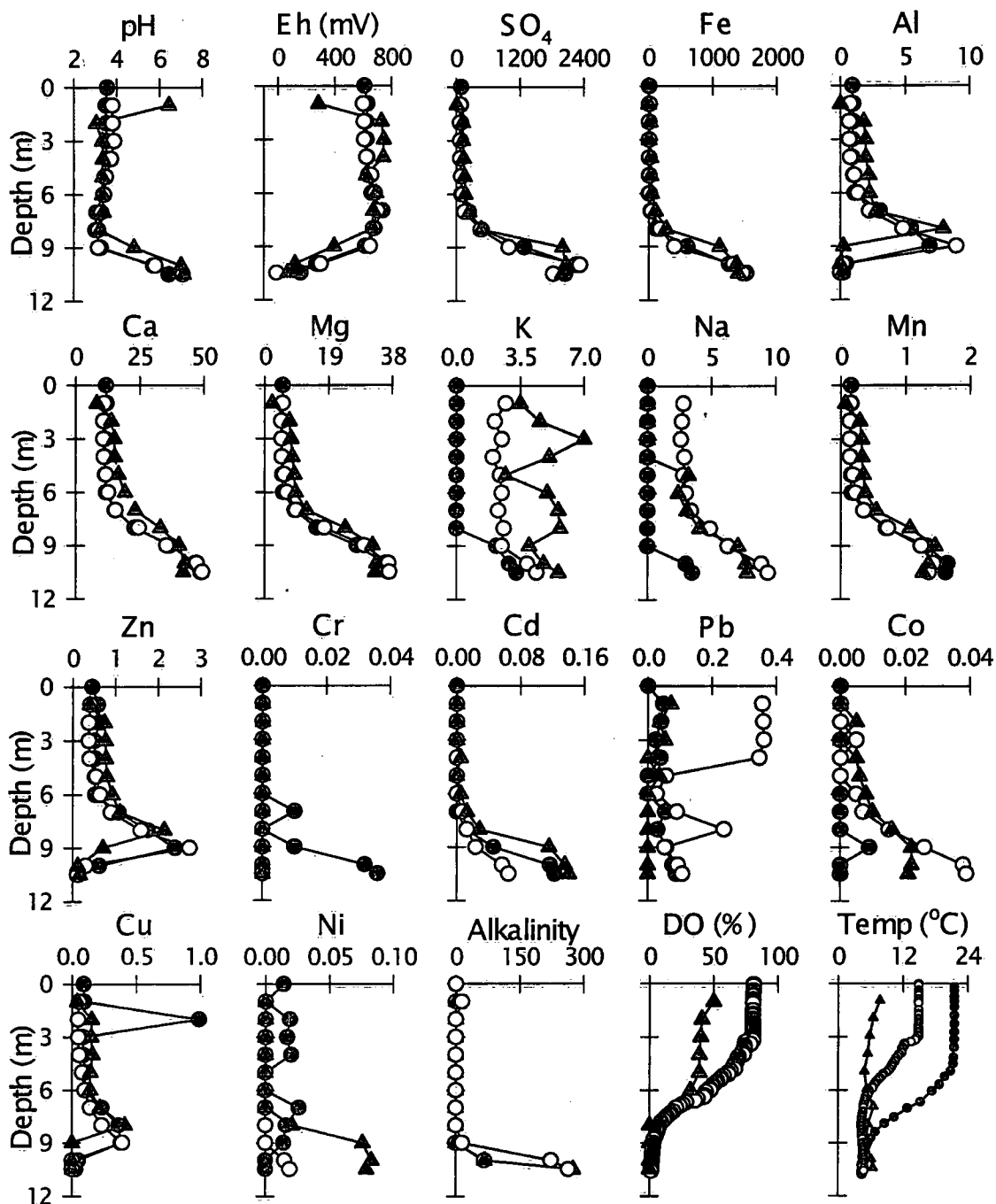


Fig.13. Depth profiles of select water chemistry in the center of the southern basin in Camp Lake (CLS1). Symbols correspond to sampling during: ● August 2001; ○ June 2002; ▲ April 2003 (beneath 1 m of ice). All values except DO and Temp are mg L⁻¹. DO refers to dissolved oxygen and alkalinity is expressed as CaCO₃. DO and alkalinity were not measured in 2001.

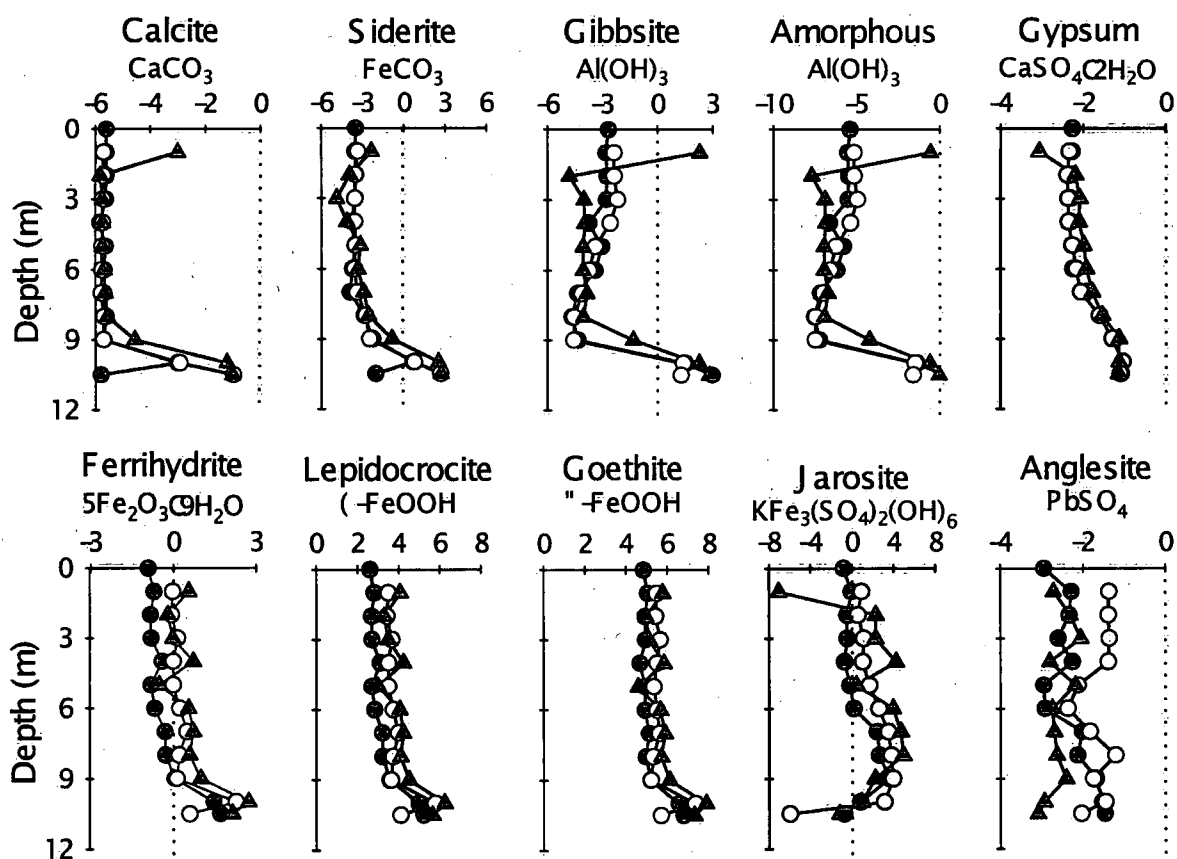


Fig. 14. Saturation indices calculated using MINTEQA2, plotted versus depth at Camp Lake (CLS1). Symbols correspond to sampling during: ● August 2001; ○ June 2002; ▲ April 2003 (beneath 1 m of ice).

Location		pH	Alk	Fe	SO ₄	Al	Zn	Cu	Ni	Cd	Co	Cr	Pb	Mn
Sherlett Creek 1985-2003	n	53	47	38	38	27	27	28	27	27	24	24	28	27
	Mean	7.24	23.6	0.16	8.87	0.05	0.13	0.01	<0.010	<0.005	<0.005	<0.005	0.05	0.02
	Max.	8.20	33.5	0.34	16.0	0.09	0.47	0.03	<0.010	<0.005	<0.005	<0.005	0.10	0.07
	Min.	6.00	5.0	0.04	4.40	<0.005	<0.005	0.002	<0.010	<0.005	<0.005	<0.005	<0.02	0.01
Woods Weir 1985-2003	n	30	3	19	19	19	19	20	19	20	18	18	20	19
	Mean	2.54	0.0	1702	4784	72.83	22.2	1.09	0.04	0.11	0.11	0.06	0.13	8.28
	Max.	3.60	0.0	2480	7500	88.0	30.1	1.37	0.07	0.16	0.15	0.09	0.28	10.70
	Min.	2.07	0.0	821	478	38.4	0.49	0.66	0.02	0.03	0.08	0.01	0.003	4.19
Acid Pond 2000-2002	n	3	3	3	3	3	3	3	3	3	3	3	3	3
	Mean	2.45	0.0	5397	16713	149	33.5	12.36	0.129	0.445	0.2033	0.088	0.613	16.3
	Max.	2.57	0.0	10900	21400	252	50.5	15.80	0.216	0.550	0.329	0.080	0.972	22.3
	Min.	2.33	0.0	3700	7150	52	18.8	8.03	0.020	0.217	0.097	0.005	0.132	6.4
Camp Weir 1983-2003	n	71	41	35	38	27	27	28	27	27	27	27	27	27
	Mean	4.12	2.0	9.10	79.5	1.21	0.54	0.10	<0.010	<0.005	<0.005	<0.005	0.156	0.17
	Max.	7.82	23.2	24.50	140	1.75	1.05	0.17	<0.010	<0.005	<0.005	<0.005	0.361	0.29
	Min.	2.91	0.0	0.14	11.2	0.45	0.05	0.01	<0.010	<0.005	<0.005	<0.005	<0.02	0.03
Drinking water concentrations	^A Max.	² 6.5-8.5		³ 0.3	³ 500	² 0.01	³ 5.0	³ 1.0		¹ 0.005		¹ 0.05	¹ 0.01	³ 0.05
Aquatic life concentrations	^B Max.	6.5-9.0		0.3		0.005	0.03	0.002-0.004	0.025-0.1	0.025-0.015		0.001-0.0089	0.001-0.007	

Data were compiled from Beck (1983), Acres (1986), unpublished data from Manitoba Conservation, and the present study

¹MAC - Maximum Acceptable Concentrations that are known or suspected to cause adverse effects on health.

²IMAC - Interim Maximum Acceptable Concentration for those substances which there are insufficient data to derive MAC, but employ a larger safety factor to compensate for the additional uncertainties involved.

³AO - Aesthetic Objectives apply to certain substances or characteristics of drinking water that can affect its acceptance by consumers or interfere with practices for supplying good quality water.

^AValues obtained from *Guidelines for Canadian Drinking Water Quality*, 2003.

^BValues obtained from *Canadian Water Quality Guidelines for the Protection of Aquatic Life*, 2002.

Table 1. Measurements of pH, alkalinity and dissolved concentrations of sulfate and metals from the long-term monitoring of Sherlett Creek, Woods Weir, Acid Pond, and Camp Weir. Alkalinity (Alk) is in mg L⁻¹ CaCO₃, Fe and all others are in mg L⁻¹. Mean refers to the geometric mean.

	Depth	Fe	Al	Zn	Cu	Ni	Pb	Cd	Co	Cr	Mn
Acid Pond	0	489	3	0.103	0.03	<2.0	0.06	<3.4	<1.0	<1.0	<1.0
CLS2	0	486	18	1.45	0.63	0.03	0.06	<3.4	0.01	0.01	0.12
CLS2	2	521	17	1.01	0.53	0.08	0.09	<3.4	0.01	0.02	0.12

Table 2. Whole-rock analyses (mg g^{-1}) of secondary Fe-oxyhydroxide precipitates from the bottom of the Acid Pond and Camp Bay (CLS2). Depth in cm refers to the sample depth interval.

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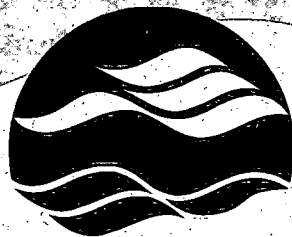
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