Treatment of Acid Rock Drainage in a Meromictic Mine Pit Lake

Timothy S. R. Fisher[1] and Gregory A. Lawrence[2]

Abstract: The Island Copper Mine pit near Port Hardy, Vancouver Island, B.C., Canada, was flooded in 1996 with seawater and capped with fresh water to form a meromictic (permanently stratified) pit lake of maximum depth 350 m and surface area 1.72 km^2. The pit lake is being developed as a treatment system for acid rock drainage. The physical structure and water quality has developed into three distinct layers: a brackish and well-mixed upper layer; a plume stirred intermediate layer; and a thermally convecting lower layer. Concentrations of dissolved metals have been maintained well below permit limits by fertilization of the surface waters. The initial mine closure plan proposed removal of heavy metals by metal–sulfide precipitation via anaerobic sulfate-reducing bacteria, once anoxic conditions were established in the intermediate and lower layers. Anoxia has been achieved in the lower layer, but oxygen consumption rates have been less than initially predicted, and anoxia has yet to be achieved in the intermediate layer. If anoxia can be permanently established in the intermediate layer then biogeochemical removal rates may be high enough that fertilization may no longer be necessary.

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Introduction

The Island Copper Mine pit lake is both a practical solution to an environmental problem and a unique limnological experiment. A meromictic (permanent density stratification) pit lake was purposefully created in 1996 by flooding the 350 m deep mine pit with seawater and capping it with fresh water (Poling et al. 2002). The pit serves as a receptacle for acid rock drainage (ARD) generated from the chemical and biological oxidation of reactive sulfide minerals in waste rock dumps. The ARD is piped to a depth of approximately 220 m and discharged through two separate diffusers as buoyant plumes. The resulting water body has evolved into three distinct layers, each of which is normally well mixed: the upper layer by wind and penetrative convection; the intermediate layer by the rising buoyant plumes; and the lower layer by geothermally driven convection.

Decommissioning of mine pits and the associated water quality problems are topical subjects that have received only limited attention in the published literature. In the broader context, the disposal of waste in stratified fluids is an important environmental issue. This paper presents a broad overview of physical and biogeochemical processes in the Island Copper Mine pit lake and their impact on the evolution of water quality. This overview is by no means complete as the pit lake is still evolving, as are strategies for its management. The paper focuses on factors that are likely to affect the long-term viability of the pit lake as a treatment system for ARD. To date concentrations of dissolved metals have been maintained well below permit limits by fertilization of the surface waters. We evaluate how dissolved metal concentrations in the upper layer will change with time under various scenarios, and examine possible management strategies to ensure that permit limits on dissolved metal concentrations will be satisfied in the long term.

Background and Study Site

Pit lakes form when abandoned mine pits fill with precipitation, groundwater, or are deliberately flooded. Depending on the mining environment, pollution in the form of ARD can be of concern, and pit lakes are often polluted with heavy metals and high acidity (Geller et al. 1998; Watkins 2000). Vertical chemical gradients are often observed and may cause meromixis. The Brenda Mines pit lake, Canada, is meromictic due to the salinity of the tailings water used to flood the pit (Stevens and Lawrence 1998; Hamblin et al. 1999), and pit lakes in Merseburg-Ost, Germany are meromictic from salt-laden groundwater inflow (Böhrer et al. 1998).

The Island Copper mine pit lake has two sharp haloclines with salinity changes of about 20%; at the upper interface and 3% at the lower interface. Conditions are similar to evaporative basins such as Mahoney Lake with salinities of 4–40% (Northcote and Hall 1983; Ward et al. 1990), but not as extreme as Mono Lake with salinities of 76–87% (Jellison et al. 1998). Anoxic basins such as Powell Lake (Sanderson et al. 1986), Nitinat Lake (Ozretich 1975), and Sakinaw Lake (Perry and Pederson 1993), all in British Columbia, and marine anoxic basins such as the Black Sea and Cariaco Trench (Millero and Sohn 1992) are related because of their seawater origins. Common to the anoxic

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[1] Senior Water Resources Engineer, Tonkin and Taylor Ltd., P.O. Box 5271, Auckland, New Zealand; formerly, Dept. of Civil Engineering, Univ. of British Columbia, Vancouver B.C., Canada V6T 1Z4. E-mail: tfisher@tonkin.co.nz.

[2] Professor, Dept. of Civil Engineering, Univ. of British Columbia, Vancouver B.C., Canada V6T 1Z4 (corresponding author). E-mail: lawrence@civil.ubc.ca

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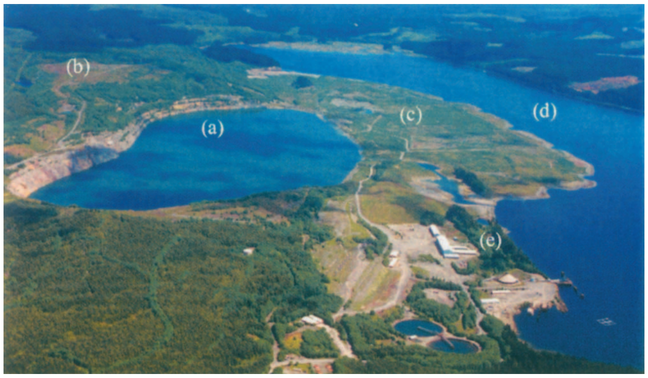

Fig. 1. (Color) Oblique aerial photograph of Island Copper Mine site and pit lake looking east. Points of interest are: (a) pit lake; (b) on land waste rock dumps; (c) beach waste rock dump; (d) Rupert Inlet; and (e) remaining mine facilities. (Photograph supplied by BHP-Billiton, with permission.)

monimolimnion (lower unmixed layer) of these systems are sulfate-reducing bacteria (SRB).

The bathymetry of pit lakes resulting from hard rock mining is characterized by high aspect ratios (Stevens and Lawrence 1998). The Island Copper mine pit was certainly deep, the bottom was 350 m below sea level making it among the lowest surface depressions on earth, and the resulting aspect ratio (depth divided by equivalent diameter) for the pit lake is 0.24. Comparable aspect ratios are found naturally only in volcanic lakes (Melack 1978; Kling 1988).

Unique to the Island Copper pit lake are buoyant plumes used for the disposal of ARD. In lakes, bubble plumes used for artificial destratification have been researched (Wüest et al. 1992; Schladow and Fisher 1995) and thermal plumes observed (Colomer et al. 2001). However, the pit lake case is more like an ocean outfall discharge (Fischer et al. 1979; Wood et al. 1993), but within a confined space like the "filling box" experiments of Baines and Turner (1969) and Kumagai (1984). In filling box experiments a plume rises within a fixed volume until the top is reached, there it spreads horizontally and downward to replace the flow entrained into the plume.

Island Copper Mine and Treatment of ARD

The Island Copper Mine, owned by BHP Billiton, is located near Port Hardy on Vancouver Island. The mine site (Fig. 1) is adjacent to Rupert Inlet, which connects to the Pacific Ocean via Quatsino Sound. Mining operations, began in October 1971 and ended in December 1995, produced mostly copper with lesser amounts of molybdenum, silver, and gold (Aspinall 1995; BHP 1996). A submarine tailings disposal system discharged mine tailings into Rupert Inlet for 23 years without significant deleterious effect to marine life (Ellis et al. 1995). Waste rock was dumped into Rupert Inlet forming the 260 ha beach dump and on land around the open pit. During the 1980s, ARD was detected from the land waste rock dumps (BHP 1996). Traces of ARD seepage from the beach dump are undetectable away from the face of the dump (Ellis et al. 1995). Tests suggest that the on-land dump material will produce moderately acidic leachate and release elevated concentrations of sulfate and metals over the long-term (Dagenais 1996).

The Island Copper Mine pit was flooded from June 15, 1996 until July 23, 1996 with seawater from Rupert Inlet, and capped with fresh water to form a meromictic lake. The resulting lake has a surface area of 172 ha, maximum depth of 350 m, average depth of 141 m, and long axis of 2.35 km (Fig. 2). The lake reached its design water level—the ultimate water level (UWL) of 2.44 m above mean sea level—during the 1998–1999 winter. Depths in the pit lake are measured as positive below the UWL.

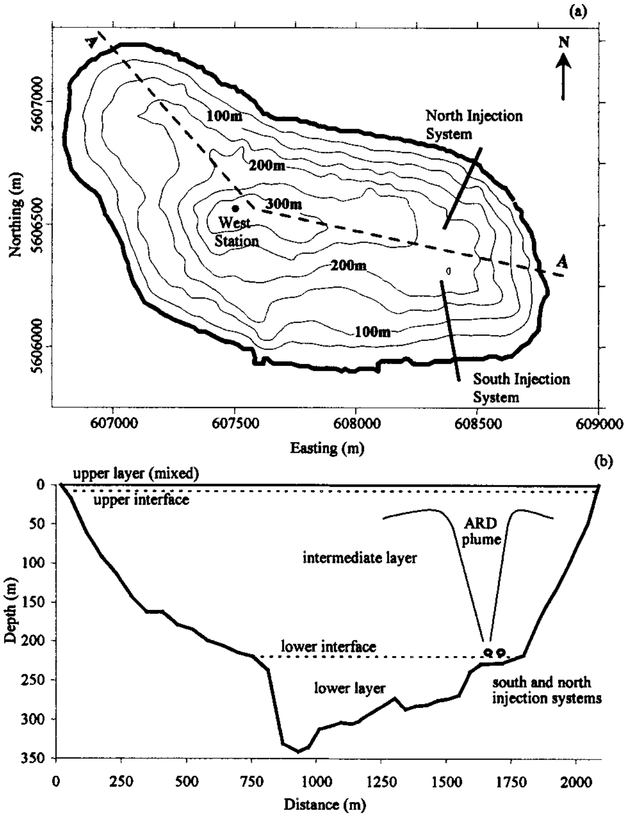

Fig. 2. Island Copper pit lake: (a) bathymetry and (b) cross-section AA. Horizontal datum is WGS-84 and depths are referenced to ultimate water level (UWL), which is 2.44 m above mean sea level (adapted from Muggli et al. 2000).

ARD collected from land waste rock dumps is discharged via two injection systems into the intermediate layer of the pit lake, where it rises as buoyant plumes through the more dense salt water (Fig. 2). The upper "brackish" water layer forms a cap over the intermediate layer, preventing the plumes from surfacing and impeding oxygen transfer to the intermediate layer. A distinct lower layer has formed below the injection system discharges. The two injection systems were installed in October 1996. The south injection system (SIS) and north injection system (NIS) drain different areas of the waste rock dump. ARD is collected by ditches, routed to sedimentation ponds, and then discharged into the pit lake by static head via two 600 mm diameter high-density-polyethylene outfall pipelines at a depth of approximately 220 m. The two diffusers consist of three sets of 200 mm ports at quarter points around the circumference, spaced along a 14 m varying diameter pipe section, and the 428 mm open end of the pipe.

An old conveyer tunnel discharges drainage into the intermediate layer at a depth of 154 m UWL. The conveyer tunnel flow is estimated to be 4% of the total injection system flow, based on a comparison of catchment areas. There are no groundwater inflows to the pit, as the pit walls are hard rock. It was only necessary to de-water the pit during mine operations to remove precipitation. The upper layer receives fresh water from direct precipitation and drainage from the pit walls. Outflow is from the surface of the lake (upper layer) through the beach dump, as a diffuse groundwater discharge into Rupert Inlet 700 m away. Initially, outflows were controlled by a concrete seepage barrier that defines the UWL. In February 1999 notches were cut in this barrier to lower the effective "weir" elevation to 0.9 m UWL (Poling et al. 2003).

The initial closure plan proposed the creation of a meromictic lake as described above, and a "passive" treatment system using removal of heavy metals by metal-sulfide precipitation via anaerobic SRB (BHP 1996), once anoxic conditions were established in the intermediate and lower layers. To accelerate anoxia, the lake has been fertilized with liquid nitrate and phosphate since 1997. This increases the primary production in the otherwise oligotrophic system, to increase organic matter in the intermediate and lower layers, which boosts the oxygen demand in these layers. Anoxia has been achieved in the lower layer, but not in the intermediate layer since oxygen consumption rates have been less than initially predicted.

Consequently, management of the pit lake has been modified. Since 2001 the less contaminated ARD from the north injection system has been discharged onto the pit lake surface. Treatment presently consists of periodic (every 7–10 days) fertilization of the surface waters to stimulate plankton blooms that absorb contaminant heavy metals, die and sink to the sediments. This treatment system has been successful in treating large quantities of ARD and producing outflow water well below effluent permit levels for all dissolved metals (Poling et al. 2003).

Methods

Field Measurements and Analysis

Conductivity–temperature–depth (CTD) profiles were made on 18 occasions: July, August, September, and November 1997; April, August, and December 1998; March, June, September, and December 1999; February, March, June, July, August, and November 2000; and March 2001. The CTD instrument, an Ocean Sensors' Model OS200 averaged 99 raw measurements every ~0.5 s. Laboratory calibration of the CTD was performed before and after each trip. Field verification of salinity was provided by bottle samples analyzed in a Guildline Model 8400 Autosal and by occasional use of supplementary CTDs. The temperature accuracy and resolution were 0.02 and 0.001°C, and for salinity they were 0.03 and 0.001%, respectively (Fisher 2002). The CTD depth uncertainty was 0.23 m based on a comparison against in situ thermistors and a Sea Bird Electronics SBE19 CTD. A depth resolution of 0.15 m was achieved across the interfaces areas by slowing the CTD descent rate.

Atmospheric temperature, humidity, shortwave radiation, wind, and precipitation were measured at a meteorological station located on the open south side of the pit adjacent to the south injection system. The wind was measured with a R.M. Young propeller vane anemometer at 4.4 m and was converted to a wind speed at a height of 10 m using the "law of the wall" assuming a neutral boundary layer (Stull 2000). Wind measurements were made for the period of December 9, 1999 to November 30, 2000. Average hourly wind speeds are used in this paper.

The above measurements have been supplemented by BHP Billiton's extensive, ongoing monitoring. Water quality measurements have been routinely taken since the pit lake was flooded. Temperature is measured by reversing thermometers and Niskin water samples taken at the West Station (Fig. 2) are analyzed for chlorinity, dissolved oxygen (modified Winkler titration), dissolved heavy metals (flame atomic absorption spectrophotometry), pH, alkalinity, and turbidity. Water level was monitored manually with a staff gauge until January 1999, and since then by an ultrasonic water level meter. Injection system inflow measurement started on November 27, 1997 using acoustic flow meters at each of the injection system outfall pipes.

Determination of Salinity and Density

The accurate field determination of salinity S (defined as the mass of dissolved salts in 1 kg of solution, expressed in parts per thousand, %) and density ρ (kg m^{-3}) is important for oceanographers and limnologists. Fortunately for oceanographers, the ionic composition of seawater varies little (Millero and Sohn 1992). There is a close relationship between the salinity and the electrical conductivity of seawater and, as this latter property can be measured easily and precisely, it is widely used to estimate salinity using the "Practical Salinity Scale 1978" (PSS78) (Perkin and Lewis 1980). The density of seawater is determined using the "Equation of State 1980" (EOS1980) as a function of temperature, salinity (PSS78), and pressure (UNESCO 1981).

While the relationship between salinity and conductivity is well understood for seawater, it is more complicated for lakes because the relative proportions of major ions vary from lake to lake, and sometimes vertically within a lake (Wuest et al. 1996). The Island Copper pit lake has the further complication that the relative proportion of major ions is changing with time, because the composition of the ARD is very different from that of seawater. For instance, while chlorine is a major ion in seawater, its concentration is negligible in the ARD. As more ARD is injected into the pit lake, the PSS78 salinity (S_{PSS}) becomes increasingly different from the actual salinity.

For the pit lake waters backcalculating salinity from density using the EOS80 was found to be the best of several methods tested for determining salinity (Fisher 2002). This approach relies on the assumption, confirmed by Wuest et al. (1996) and Millero (2000), that changing chemical composition does not adversely affect the equation of state. This assumption was further confirmed for the Island Copper pit lake waters using samples of upper layer (5 m), intermediate layer (100 m), and lower layer (300 and 350 m) water collected on November 29, 2000 (Fisher 2002). The densities of the sample were accurately determined using an Anton Parr densitometer. Following Millero (2000) we treat the density derived salinity as the "true salinity" and relate it to the S_{PSS} by

$$S = S_{\mathrm{PSS}} + (t/1{,}590)\Delta S \qquad (1)$$

where ΔS=empirically determined layer dependent salinity correction. The time (t, units days) dependency assumes a linear change in salinity from the creation date of the lake (July 23, 1996) until when the analysis sample was taken (November 29, 2000, t=1,590 days). The salinity corrections for the upper, intermediate and lower layers are

Upper

$$\Delta S = 0.0359 - 0.00329\,T + 0.0525\,S_{\mathrm{PSS}} \quad \text{for } 4 < T < 20,\ 3 < S_{\mathrm{PSS}} < 6 \qquad (2a)$$

Intermediate

$$\Delta S = -0.182 - 0.00530\,T + 0.0204\,S_{\mathrm{PSS}} \quad \text{for } 11 < T < 13,\ 20 < S_{\mathrm{PSS}} < 27 \qquad (2b)$$

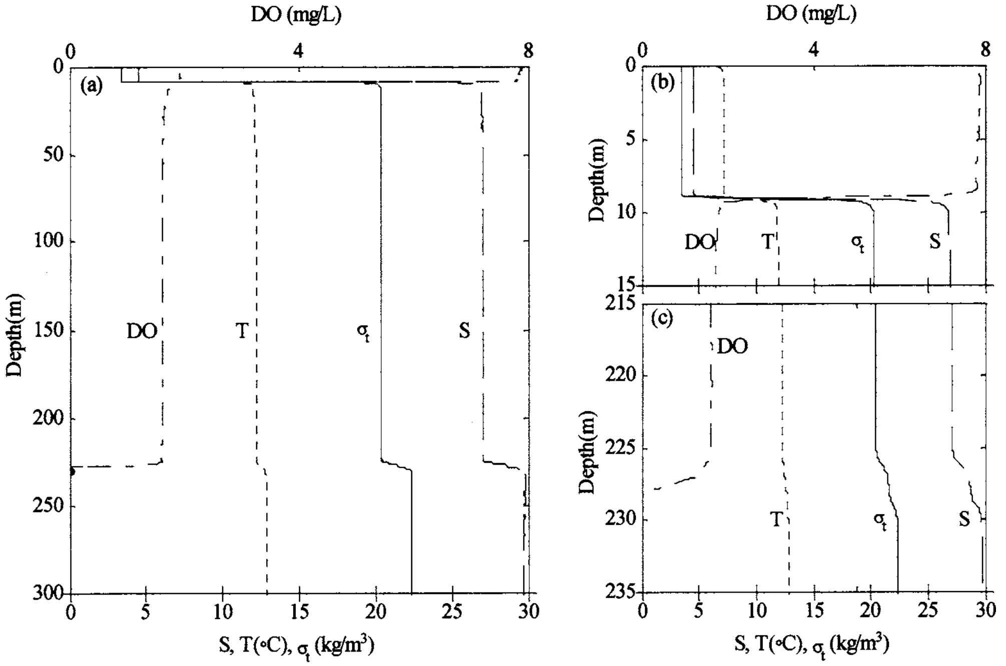

Fig. 3. Vertical profiles of: (a) full depth; (b) upper interface; and (c) lower interface showing salinity (%), temperature, density as $\sigma_t = \rho(S, T, P=0)$-1,000 and dissolved oxygen (DO) for December 6, 1999

Lower

$$\Delta S = 0.0420 + 0.00206\ T + 0.0447\ S_{\mathrm{PSS}} \quad \text{for } 12 < T < 14,\ 27 < S_{\mathrm{PSS}} < 30 \qquad (2c)$$

where T=temperature (°C). Eqs. (1) and (2) provide the necessary correction to S_{PSS} to enable density to be computed from CTD field data. Applying Eqs. (1) and (2) for November 29, 2000 pit lake conditions, the salinity correction was 0.23, 0.29, and 1.34% for the upper, intermediate, and lower layers, respectively.

The first CTD profile was in July 1997. Prior to that date the only measure of salinity was the chlorinity, determined by silver nitrate titration. The chlorinity salinity, $S_{\mathrm{Cl}} = 1.80655$ (Cl%) serves as the best estimate of salinity before July 1997. It provides an estimate of the "seawater salinity," the contribution to salinity from the original seawater.

Results

Layers

The pit lake has evolved into a stably stratified three-layer system. The CTD profile taken December 6, 1999 is typical of other dates, with salinity, temperature, density, and dissolved oxygen being approximately constant in each layer (Fig. 3). The layers were separated by haloclines at ~8 and ~225 m, respectively. The upper layer volume, $V_1 = 0.134 \times 10^8$ m^3 (5.5% of the total lake volume); the intermediate layer volume, $V_2 = 2.17 \times 10^8$ m^3 (88.5%), and the lower layer volume, $V_3 = 0.148 \times 10^8$ m^3 (6.0%).

ARD

ARD flow to the pit lake via the NIS and SIS has been highly seasonal. Flows increase through the fall, reach a peak in December and/or January and then drop through the spring to minimal summer values. The annual daily average flows [and peak daily flow] for the NIS and SIS over the period December 1997–November 2000 were 7,190 m^3/day (75,200 m^3/day) and 4,650 m^3/day (72,000 m^3/day), respectively. The ARD salinity (noncorrected PSS78) measured in the SIS pond over the same period had a mean of 0.82% and a standard deviation of 0.28%.

Salinity

The pit was initially flooded with seawater from Rupert Inlet. A brackish upper layer formed on the surface of the lake due to precipitation, runoff, and additional fresh water (pumped from the nearby Marble River until March 15, 1998). The salinity of the upper layer decreased rapidly at first, due to dilution from the Marble River supply, and more slowly since March 15, 1998 reaching 4.08% in March 2001 [Fig. 4(a)]. There is a slight seasonal variation in upper layer salinity. Winter precipitation dilutes the upper layer reducing the salinity, whereas, during summer (when there is relatively little precipitation) mixing across the upper halocline of intermediate layer water causes the upper layer salinity to increase slightly.

As a result of the relatively fresh ARD discharge, the intermediate layer salinity decreased from an initial value of 28.87% to 26.62% in March 2001 [Fig. 4(b)]. The most rapid decreases occurred in the winter when injection system flows were the greatest. Diversion of the NIS from the intermediate layer to upper layer on June 29, 2000 reduced the dilution of the intermediate layer salinity during the 2000–2001 winter.

The lower layer salinity increased to 29.78% in March 2001, which is greater than the initial seawater salinity [Fig. 4(b)]. The seawater salinity of the lower layer, which represents the concentration of original seawater ions, was diluted over this period indicating an exchange of fluid between the intermediate and lower layers. In all layers there is a growing difference between the true and seawater salinity due to the addition of nonseawater ions, which are primarily ions dissociated from the pit walls and from ARD (Fisher 2002). In the lower layer the concentration of

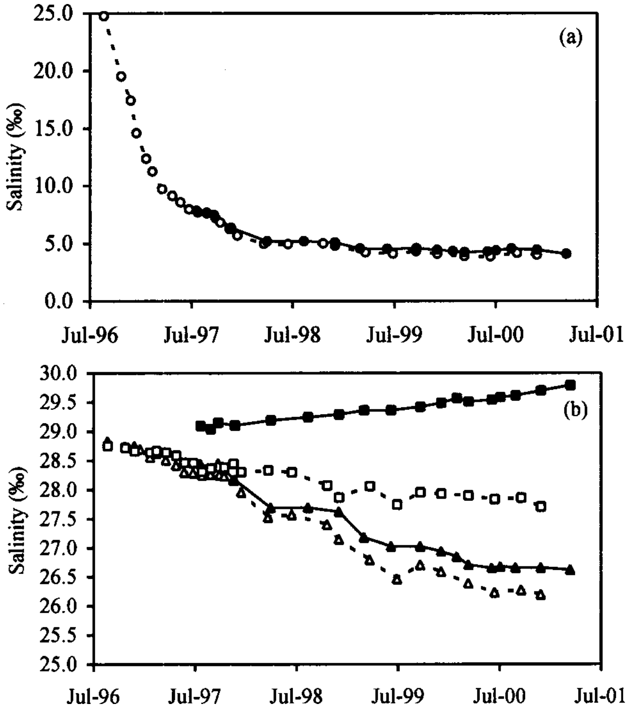

Fig. 4. Salinity (bold symbols) and seawater salinity (open symbols) of: (a) upper layer [●,○]; and (b) intermediate layer [▲,△] and lower layer [■,□]

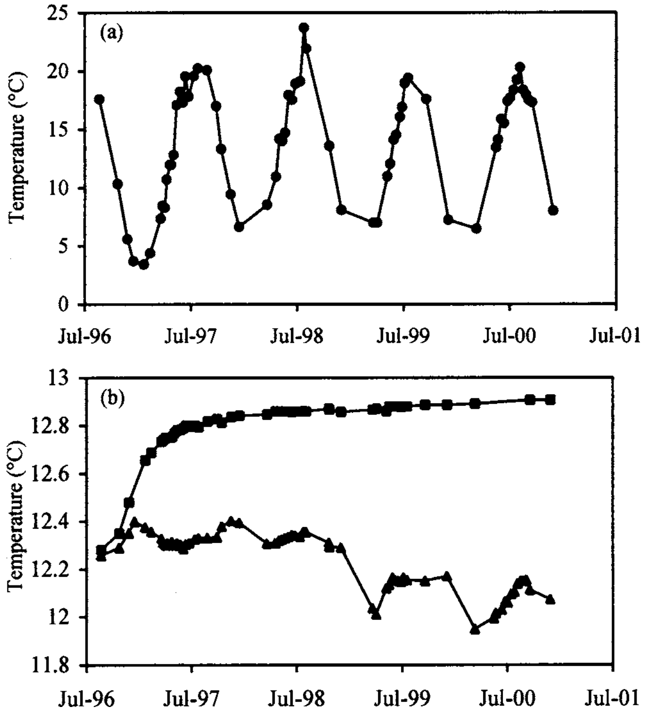

Fig. 5. Temperature of: (a) upper layer [●]; and (b) intermediate layer [▲], and lower layer [■]

nonseawater ions is enhanced by lower pH, smaller volume to surface area ratio, and the greater availability of loose rock (Fisher 2002). In addition, the lower layer may be more biochemically active due to anoxia and accumulation of biomass from the upper layers.

Temperature

The temperature of the upper layer followed the seasonal cycle of air temperature with summer highs up to 24°C and winter lows down to 4°C [Fig. 5(a)]. The intermediate and lower layers remained between 12 and 13°C [Fig. 5(b)]. The temperature of the intermediate layer was generally decreasing, with seasonal variations due to winter influx of cold ARD and summer solar heating. The temperature in the lower layer rose to 12.9°C, presumably as a result of geothermal heating [Fig. 5(b)].

Dissolved Oxygen

The dissolved oxygen (DO) concentration of the upper layer has remained near saturation and fluctuated seasonally with water temperature (Fig. 6). The intermediate and lower layer DO concentrations decreased to 2.48 and 0.03 mg/L, respectively, by November 29, 2000. The lower layer was essentially anoxic as the DO concentration was at the detection limit. The intermediate layer had a significant amount of oxygen with only a slow rate of reduction.

pH

The pH of the upper layer fluctuated seasonally between 6.2 and 8.8. The variations in the pH of the intermediate and lower layers were not so regular, exhibiting mean (and standard deviation) values of 6.63 (0.15) and 6.31 (0.16), respectively, since January 1997. The pH of the NIS and SIS flows were 7.1–8.4 and 4.2–6.6, respectively.

Copper and Zinc

Copper and zinc concentration increased with depth (Fig. 7). Zinc concentrations in the upper layer decreased each summer coinciding with fertilization of the pit lake. Decreases in copper concentrations also coincided with fertilization, but the trend was weaker and was obscured by measurement resolution. Fertilization was found to correlate well with increased chlorophyll "*a*" levels in the upper layer and decreased seiche depths (Poling et al. 2003). Delaying the start of the fertilization program in 2000 until mid-August proved the dependence of these trends on fertilization, rather than increased summer solar radiation. The changes to chlorophyll *a* levels, secchi depths, and metal concentrations (Fig. 7) were all correspondingly delayed until the start of

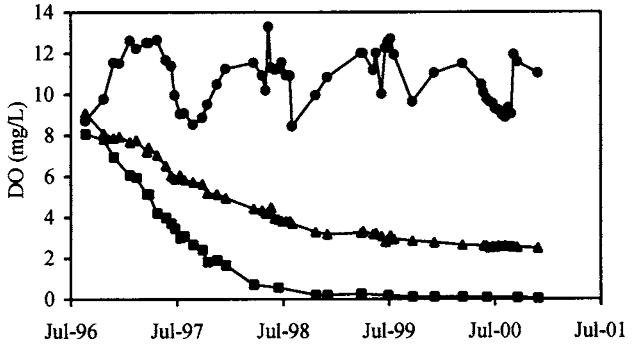

Fig. 6. Dissolved oxygen of upper layer [●], intermediate layer [▲], and lower layer [■]

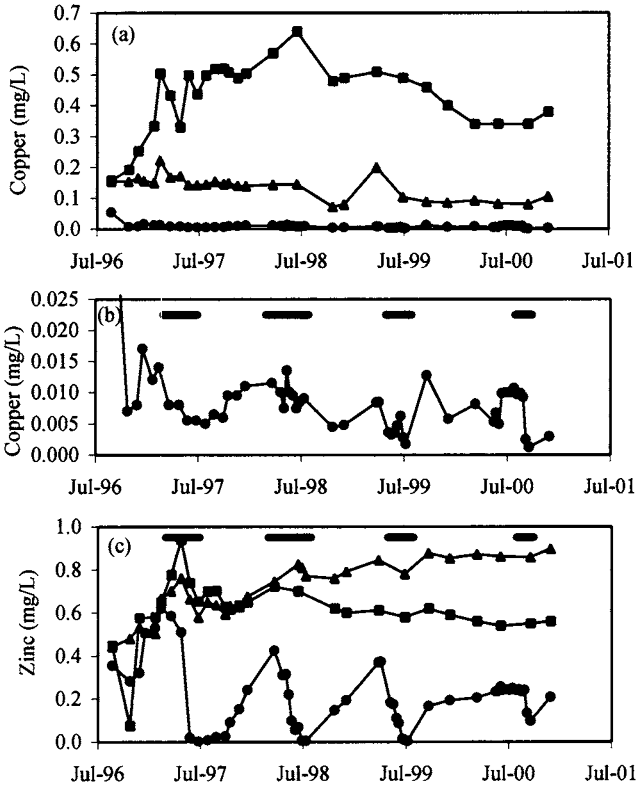

Fig. 7. (a) Copper, (b) copper (upper layer only), and (c) zinc concentration of upper layer [●], intermediate layer [▲], and lower layer [■], with [—] indicating fertilization periods

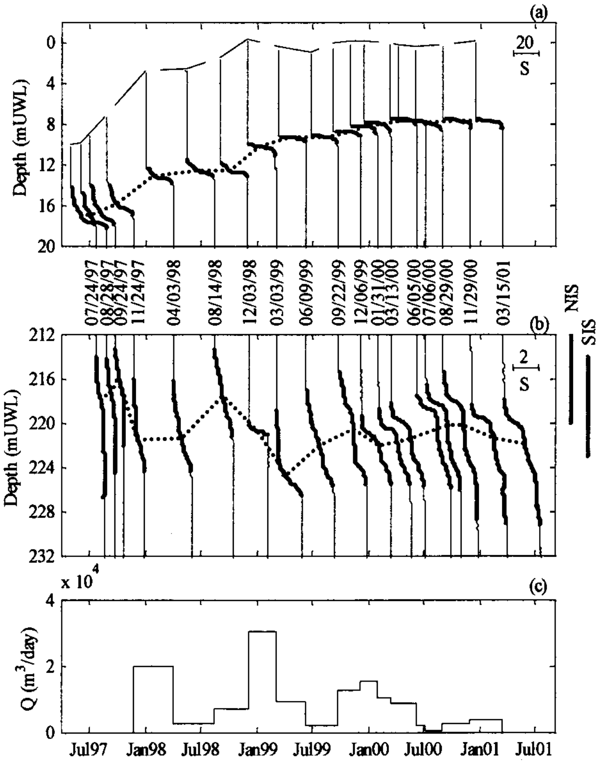

Fig. 8. Time series of salinity (%) profiles through: (a) upper halocline and (b) lower halocline with surface water level [– –] and effective halocline depth [---]. Thicker lines represent extent of halocline. Bars adjacent to (b) indicate depths of north injection system (NIS) and south injection system (SIS). (c) Total injection system flow averaged over period between CTD casts.

fertilization. The decrease in Zn and Cu in the upper layer is attributed to biological removal by phytoplankton by adsorption or active uptake (Poling et al. 2003).

Upper Halocline and Upper Layer

The upper halocline rose 10.2 m from 18.05 m UWL in July 1997 to 7.85 m UWL in March 2001 [Fig. 8(a)]. Initially the halocline rose relatively rapidly due to the injection of ARD into the intermediate layer. However, with the filling of the pit lake in February 1999, and the subsequent discharge of water through the beach dump, the upper layer has thinned. With a thinner upper layer, mixing from the wind and penetrative convection erodes the halocline from above and limits its rise—it even fell slightly during the summer of 2000. The halocline is also sharpened from below by the action of the ARD plumes. The result is that the halocline is less than 1 m thick, which is remarkable given the salinity jump of more than 20%.

Intermediate Layer

The discharge of ARD into the intermediate layer results in slight temperature, salinity, and density variations of order 0.05%, 0.05°C, and 0.05 kg/m^3, respectively, over the depth of the layer. The density gradients are strongest in winter when the ARD discharge rate is highest. These observations are consistent with the "filling box" mechanism, whereby a buoyant plume discharged into a confined region causes the receiving environment to become stratified (Baines and Turner 1969). While these gradients are small they can be important. During low injection system flows the ARD plume was observed by dye tracer experiments to be trapped at depth by the intermediate layer density gradient, and for higher flows to rise to the upper halocline before spreading horizontally (Muggli et al. 2000). These dye tracer experiments were performed on the plumes in October 1998 at the NIS for a flow 1,600 m^3/day (22% of NIS average), and in March 1999 at the SIS for a flow of 10,800 m^3/day (230% of SIS average). At the low flow the plume rose to a depth of 90 m UWL and slowly spread in a 10 m deep layer horizontally across the lake. At the higher flow the plume rose rapidly to the upper halocline, and then spread radially in a 60 m deep layer reaching the pit walls within 24 h.

Lower Halocline and Lower Layer

The lower halocline had an average thickness of 8.1 m and has a seasonal oscillation of up to 9 m [Fig. 8(b)]. It deepens in the fall/winter and rebounds in the spring/summer. The halocline deepened considerably in the winters of 1997–1998 and 1998–1999, but less so for the winters of 1999–2000 and 2000–2001, corresponding to the observed variation in ARD flow [Fig. 8(c)]. The unusual shape of the lower halocline, particularly the observed double step is probably due to the different depths [Fig. 8(b)] and variable flows of the NIS and SIS diffusers. Inspection of the injection systems by a submersible remotely operated vehicle in April 2001 revealed that both diffusers were lying on the pit walls at an angle of 37° from horizontal (Rescan, personal communication, 2001). The SIS diffuser has only three ports open (out of 12) plus the end of pipe. Furthermore, both diffusers were only discharging from the first set of diffuser ports,

Table 1. Data Used in Pit-Lake Model

	Concentrations (mg/L)						Volumetric inflow rates[a] (10^6 m^3/year)					Mass fluxes (t/year)			
	C_S[b]	C_N[b]	C_R[c]	C_{1i}[d]	C_{2i}[d]	C_{PER}	Q_S	Q_N	Q_R	Q_P	Q_1	M_S	M_N	M_R	M
Zn	5.93	1.88	3.51	0.002	0.7	1.0	1.75	2.63	0.50	3.26	8.14	10.4	4.94	1.75	17.1
Cu	0.79	0.02	0.33	0.003	0.06	0.05						1.38	0.05	0.16	1.60

[a] From Poling et al. (2003).

[b] Average of 1998–2000 concentrations.

[c] Average of samples taken from five sites around the pit walls.

[d] Representative 2002 values from Poling et al. (2003).

indicating that at that time the intrusion of saline pit lake water was causing the diffusers to be partially blocked (Wilkinson 1984). The diffuser ports are spaced around the circumference of the diffuser pipe. So the ARD discharge is in fact a series of plumes discharging at different elevations, at different angles to the horizontal and the vertical, and at different rates and times. Therefore, it is not surprising that the halocline has a complicated structure.

At the lower halocline, the large seasonal oscillation is believed to result from the balance of mixing energies across the halocline. The large depth changes of the lower halocline, compared to the upper halocline, result from smaller density difference rather than stronger mixing processes. Turner (1973) summarized the effect of entrainment as being the transport of fluid from the quiescent fluid into the stirred fluid. Thus, for deepening of the lower halocline (or upward entrainment of fluid) there must be greater stirring in the intermediate layer than the lower layer. Indeed, the intermediate layer is strongly stirred by the injection system plumes (Muggli et al. 2000; Rescan, personal communication, 2001). Rapid deepening of the halocline is correlated with periods of high ARD flow (Fig. 8).

The observed summer rise of the lower halocline (and downward entrainment of fluid) implies that the lower layer is stirred more vigorously than the intermediate layer during periods of low flow. The cause of the mixing in the lower layer can be deduced from Fig. 5. The lower layer is steadily warming and the intermediate layer is cooling from year to year, with seasonal fluctuations. In 2001 the temperature difference between the two layers approached 1°C. Mixing of the lower layer may be analogous to the classic problem of thermal convection, where fluid between two plates is heated from below and cooled from above. In the case of the lower layer heating from below is provided by geothermal heating and cooling from above by the cooler intermediate layer. The heat flux into the lower layer was initially ~ 8 W/m^2, but has subsequently dropped to 0.23 W/m^2, which is still high compared to the geothermal heat fluxes (0.02–0.04 W m^{-2}) observed in B.C. Fjords (Hyndman 1976). These rates were calculated based on the observed temperature increase [Fig. 5(b)] and represent lower bounds as they do not include heat lost across the lower halocline. Note the initial high rate of heating is attributed to the gradual release of heat stored in the pit walls from solar radiation prior to flooding, but cannot be confirmed. Confirmation of the erosion of the lower halocline by active mixing processes in the lower layer is provided by the dilution in seawater salinity of the lower layer [Fig. 4(b)], which can only result from transport of intermediate layer fluid with less seawater salinity to the lower layer.

The conditions are right for double diffusion across the lower halocline, as warm saline fluid underlies colder fresher fluid. However, the step-like structures observed at the lower halocline are not sharp enough to be double diffusive, but rather are believed to be formed by the uneven erosion of the lower halocline by the NIS and SIS diffusers.

Discussion

The initial mine closure plan proposed removal of heavy metals by metal–sulfide precipitation via anaerobic SRB (BHP 1996), once anoxic conditions were established in the intermediate and lower layers. By 2000 it became apparent that while anoxia was imminent in the lower layer (Fig. 6), without any changes to the management of the lake the prospect of anoxia in the intermediate layer within 10 years was remote (Poling et al. 2003). Consequently, in June 2000, the treatment system was modified to both decrease the oxygen input and increase oxygen consumption in the intermediate layer, by redirecting the less contaminated NIS discharge (see Table 1) to the pit lake surface. Since May 2001 the surface waters of the lake have been successfully treated by fertilization every 7–10 days to stimulate plankton blooms that absorb contaminant heavy metals, die and sink to the sediments. Poling et al. (2003) report that green algae dominates the phytoplankton assemblage. In this discussion calculations will be presented to assess the implications of recent modifications to the treatment system, as well as the possibility of eventually establishing a truly passive treatment system.

Dissolved Oxygen

The initial rate of oxygen reduction in the lower layer was much greater than in the intermediate layer. For example in 1997 the DO in the lower layer dropped by approximately 5 mg/L (0.014 mg/L/day), whereas in the intermediate layer it only dropped by 3 mg/L (0.008 mg/L/day). Possible reasons for the difference are the settling of organic debris into the lower layer, a relatively greater sediment oxygen demand in the lower layer, and the direct input of DO into the intermediate layer via the ARD injection. In addition some DO may be mixed downward into the intermediate layer across the upper halocline by the action of the ARD plumes, but this is likely to be only a small fraction of the direct injection of DO (Fisher 2002).

The rate of increase in DO within the intermediate layer due to the ARD inflow is

$$I_2 = \frac{Q_A \, \mathrm{DO}_A}{V_2} \tag{3}$$

where V_2=volume of the intermediate layer; DO_A=DO of the ARD; and Q_A=total ARD discharge rate. If representative average values are used, then

$$I_2 \approx \frac{1.2 \times 10^4 \text{ m}^3\text{/day} \times 11 \text{ mg/L}}{2.17 \times 10^8 \text{ m}^3} \approx 0.0006 \text{ mg/L/day}$$

So, initially at least, the injection of DO into the intermediate layer was dwarfed by consumption. If the initial consumption rate had been sustained then the intermediate layer would have become anoxic in 1999. However, during 1998 and 1999 the consumption rate dropped dramatically and by June 2000 the DO concentration was still 2.6 mg/L, and dropping at a rate of about 0.0008 mg/L/day, an order of magnitude slower than the initial rate.

Diverting the NIS to the upper layer in June 2000 did not have a significant impact on the DO concentration in the intermediate layer (Fig. 6). The reduction in oxygen input was balanced, at least approximately, by the continuing decrease in consumption. To increase consumption year round fertilization was commenced in May 2001, and since that time the DO concentration has decreased more rapidly, and by December 2004 had dropped to less than 0.4 mg/L (Rescan, personal communication). It now appears likely that if year-round fertilization is continued anoxia will be achieved in the near future.

Upper Layer Thickness

Why not return the NIS discharge to the intermediate layer, since its impact on DO is not dramatic? In fact the NIS discharge was returned to the intermediate layer from September 2001 until August 2002, during which time the upper layer thinned from 7 to about 4 m. This thinning caused concern as to the future of the upper layer, and the NIS was rediverted to the upper layer in August 2002 (Poling et al. 2003). Subsequently, the minimum upper layer thickness has been 3.4 m (Rescan, personal communication). The upper layer thickness varies as the elevation of the free surface and the upper halocline vary. The free surface level is determined by the total inflow into the pit lake and the outflow from the upper layer, which flows over a concrete seepage barrier and through the porous beach dump into Rupert Inlet. The upper halocline elevation is determined by a number of factors including: the rate at which ARD is injected into the intermediate layer [the average NIS (SIS) discharge rate corresponds to a 1.5 m (1.0 m) rise per year]; downward mixing due to wind and penetrative convection; and downward mixing due to the NIS discharge, depending on if, and how, it is injected into the upper layer.

An additional consideration is the possibility that a sufficiently strong wind will tilt the halocline, such that the halocline and intermediate layer water may rise to the surface at the upwind end of the lake (upwelling). This would greatly enhance mixing between the upper and the intermediate layers and potentially lead to destratification of the upper and the intermediate layers. The Wedderburn number

$$W = \frac{g' h_1^2}{u_*^2 L} \tag{4}$$

is an indicator of the likelihood of upwelling ($W < 1$) for a two layer system (Monismith 1985). The stabilizing effect of buoyancy is parametrized by the upper layer depth, h_1, and the reduced gravitational acceleration, $g' = (\rho_2 - \rho_1) g / \rho_2$, where g=gravity; and ρ_1 and ρ_2=densities of the upper and intermediate layers, respectively. The destabilizing effect of wind is parametrized by the lake length, $L = 2.35$ km, and shear velocity, $u_* = \sqrt{C_D \rho_a U_{10}^2 / \rho_1}$, where $C_D \approx 0.0013$=drag coefficient; $\rho_a \approx 1.25$ kg/m^3=air density; and U_{10}=wind-speed 10 m above the water. From Eq. (4) we can calculate a critical upper layer depth

$$h_{1c} = 0.062 \frac{U_{10}}{\sqrt{g'}} \tag{5}$$

representing the minimum thickness of the upper layer needed to ensure that $W < 1$. Direct measurements on March 15, 2001 gave $\rho_1 = 1.4$ kg/m^3 and $\rho_2 = 18.4$ kg/m^3, and $g' = 0.16$ m/s^2. Analysis of wind records from the site and nearby Port Hardy yields a 1 in 10 year, 1 h wind speed of 17.6 m/s (Fisher 2002). Substituting these values into Eq. (5) gives $h_{1c} = 2.7$ m, a value that will increase with time as more ARD is injected into the intermediate layer, reducing its density and g'. Thus, not only is diverting the NIS to the upper layer desirable to reduce the rate at which the upper layer thins; it will help maintain the density difference between the layers, and reduce the rate at which h_{1c} increases. Note that it is not uncommon for the free surface level to be 2 m above the effective weir height, increasing the upper layer thickness needed to prevent the flow of intermediate layer water to flow over the seepage barrier.

The level of the upper halocline needs to be managed to prevent upwelling. An option would be to make use of the kinetic energy of the NIS to entrain intermediate layer fluid into the upper layer as the halocline starts to rise above a predetermined elevation. Given that the NIS flow is 1.5 times the SIS flow, this could easily be achieved. In fact it is possible that only a small portion of the NIS would need to be diverted, since diffusers are routinely engineered to entrain many times their own flow rate (Fischer et al. 1979; Wood et al. 1993).

Metal Concentrations

Both the results presented in Fig. 7 and those in Poling et al. (2003) demonstrate the effectiveness of fertilization in maintaining upper layer concentrations of Zn and Cu well below the effluent permit limits ($\text{Zn}_{\text{PER}} = 1.0$ mg/L and $\text{Cu}_{\text{PER}} = 0.05$ mg/L) applicable to the upper layer of the pit lake for compliance with government regulations (Poling et al. 2002). In addition to fertilization Poling et al. (2003) discuss three other possible biogeochemical removal processes. First, neutralization of the more acidic and highly contaminated SIS discharge results in an abundance of white precipitate (aluminum and iron oxy-hydroxides) that may scavenge heavy metals by adsorption. Second, the cycling of manganese and iron between the sediment and the water column may transform dissolved heavy metals into insoluble sulfides. Finally, sulfide precipitation by SRB will occur under anoxic conditions. However, if fertilization is discontinued, and these other biogeochemical removal processes are not active, then the upper layer concentrations of Zn and Cu could rise above the permit levels.

To illustrate the possibility of excessive metal concentrations in the upper layer we use the pit-lake model depicted in Fig. 9. Unless otherwise stated the model assumes that the NIS is diverted to the upper layer. In this model the upper and intermediate layers are treated as continuously stirred reactors satisfying

$$\frac{d(C_j V_j)}{dt} = M_j - C_j Q_j - R_j \tag{6}$$

where $j = 1$ refers to the upper layer and $j = 2$ refers to the intermediate layer (we do not need to consider the lower layer in the calculations that follow). The dissolved concentration of the metal

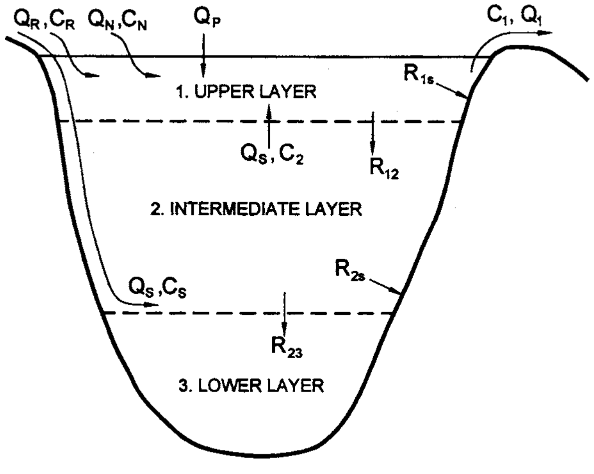

Fig. 9. Schematic of pit lake showing flow rates, concentrations, and removal rates

under consideration is C_j. The layer volumes, V_j, and flow through rates, Q_j, are assumed constant. There are two important time scales: the retention time of the upper layer, $t_{R1}=V_1/Q_1 \approx 1.6$ years, and the retention time of the intermediate layer, $t_{R2}=V_2/Q_2 \approx 124$ years (50 years if NIS injected into intermediate layer).

Each of the three terms on the right hand side of Eq. (6) represents a mass flux of the dissolved metal under consideration (Cu or Zn). The first term, M, is the flux of dissolved metal into a layer. For the configuration depicted in Fig. 9, $M_1=M_N+M_R+C_2Q_S$, and $M_2=M_S$, where $M_X=C_XQ_X$, and the subscripts S, N, R, and P represent the SIS and NIS discharges, the local runoff from the pit walls, and direct precipitation onto the pit, respectively. The second term, C_jQ_j, is the flux of dissolved metal out of a layer, where $Q_1=Q_S+Q_N+Q_R+Q_P$, and $Q_2=Q_S$. Measured values of the above concentrations and inflow rates are presented in Table 1.

The last term, R_j, is the net flux of metal removed from a layer by physical and biogeochemical processes. The removal may be due to by settling into a lower layer or onto the sediments. The upper layer removal rate, $R_1=R_{12}+R_{1s}$, and the lower layer removal rate, $R_2=R_{23}+R_{2s}-R_{12}$, where R_{12}, R_{23}, R_{1s}, and R_{2s} are the removal rates from the upper to the intermediate layer, from the intermediate to the lower layer, and from the upper and intermediate layers to the sediments, respectively (Fig. 9). For simplicity we assume constant removal rates.

Solving Eq. (6) for the concentration in the intermediate later yields

$$C_2 = C_{2\infty} + (C_{2i} - C_{2\infty})e^{-\tau_2} \tag{7a}$$

where $\tau_2=t/t_{R2}$; and C_{2i}=initial concentration in the intermediate layer. The asymptotic concentration in the intermediate layer is given by

$$C_{2\infty} = \frac{(M_2 - R_2)}{Q_2}, \quad \text{providing } R_2 \leqslant M_2 \tag{7b}$$

Note, if $R_2 > M_2$, Eq. (6) applies until $C_{2\infty}$ drops to zero.

Solving Eq. (6) for the concentration in the upper layer yields

Table 2. Copper and Zinc Concentrations for Various Removal Rate Scenarios

	Removal rates (t/year)		Concentrations (mg/L)			
Scenario	R_2(Cu)	R_2(Zn)	$\mathrm{Cu}_{1\infty}$	$\mathrm{Cu}_{1\,\max}$	$\mathrm{Zn}_{1\infty}$	$\mathrm{Zn}_{1\,\max}$
1	0	0	0.18	0.18	2.10	2.10
2	0.69	5.2	0.098	0.098	1.46	1.46
3	1.38	10.4	0.013	0.025	0.82	0.96
4	1.44	15.3	0.0068	0.037	0.22	0.57

$$C_1 = C_{1\infty} + (C_{1i} - C_{1\infty})e^{-\tau_1} + (C_{2i} - C_{2\infty})\left(\frac{Q_2}{Q_1}\right)\left[\frac{e^{-\tau_2} - e^{-\tau_1}}{1 - (\tau_2/\tau_1)}\right] \tag{8a}$$

where $\tau_1=t/t_{R1}$; and C_{1i}=initial concentration in the upper layer. The asymptotic concentration for the upper layer

$$C_{1\infty} = \frac{M - R}{Q_1} \quad \text{providing } R \leqslant M \tag{8b}$$

where $M=M_S+M_N+M_R$ and $R=R_1+R_2$.

If the removal rates are set to zero then $\mathrm{Zn}_{1\infty}=2.1$ mg/L and $\mathrm{Cu}_{1\infty}=0.18$ mg/L, both of which are significantly greater than the permitted levels, therefore some removal is needed. Note, from Eq. (8*b*), that for the asymptotic concentration to be less than the permit concentration

$$R > R_{\mathrm{crit}} = M - C_{\mathrm{PER}}Q_1 \tag{9}$$

Substituting the values from Table 1 into Eq. (9) yields critical removal rates $R_{\mathrm{crit}}=1.2$ and 9.0 t/year for Cu and Zn, respectively. Such removal rates are achievable, in fact Poling et al. (2003) report $R \geqslant 27$ t of Zn during intensive fertilization from September 2001 to September 2002.

An important question is: could permit limits be satisfied if fertilization were discontinued? Conceivably, once anoxia is established in the intermediate layer sulphide precipitation by SRB, together with other biogeochemical removal processes, could remove metals at a sufficiently rapid rate to ensure that concentrations in the upper layer never exceed the permit limits. We will consider four scenarios with different rates of removal from the intermediate layer, but no removal from the upper layer ($R_1=0$):

1. No biogeochemical removal from the intermediate layer ($R_2=0$);
2. Biogeochemical removal from the intermediate layer at 50% of the rate at which the SIS delivers metals to the intermediate layer ($R_2=0.5\ M_S$);
3. Biogeochemical removal from the intermediate layer at the same rate as the SIS delivers metals to the intermediate layer ($R_2=M_S$); and
4. Redirection of the NIS to the intermediate layer, and removal equal to the sum of the rates at which the SIS and NIS deliver metals to the intermediate layer ($R_2=M_S+M_N$).

The asymptotic concentrations for the upper layers under each of these scenarios are presented in Table 2. With no removal (Scenario 1) the asymptotic concentrations exceed the permit levels in both layers. The asymptotic concentrations in the intermediate layer are much higher than those in the upper layer. Under Scenario 2 ($R_2=0.5\ M_S$) the asymptotic concentrations in the upper layer are reduced dramatically, but they are still greater than the permit limits, since the removal rates (Table 2) are less

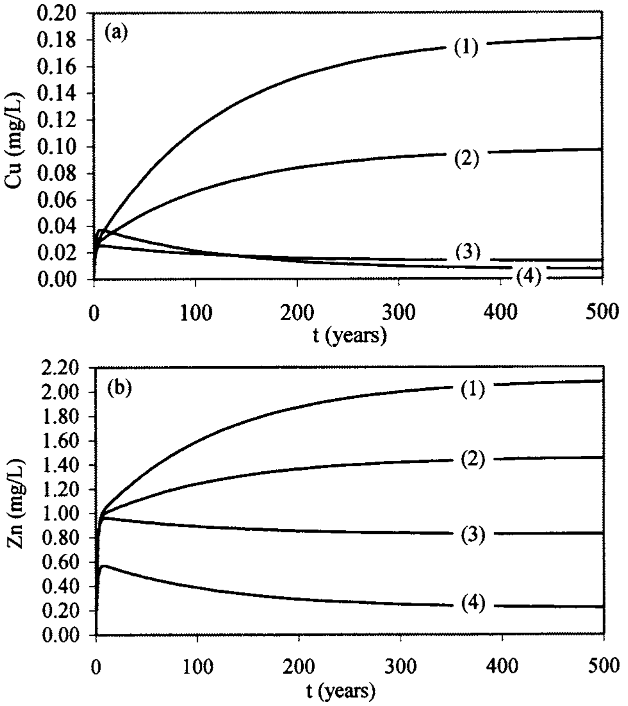

Fig. 10. Predicted time evolution of: (a) copper and (b) zinc concentrations in the upper layer under four removal rate scenarios: (1) $R_1=0, R_2=0$; (2) $R_1=0, R_2=0.5\,M_S$; (3) $R_1=0, R_2=M_S$; and (4) $R_1=0, R_2=M_S+M_N$

than the critical removal rates. However, under Scenario 3 ($R_2=M_S$) the critical removal rates for both Cu and Zn are exceeded, and the asymptotic concentrations in the upper layer are less than the permit limits. The asymptotic concentrations are reduced even further under Scenario 4 ($R_2=M_S+M_N$), particularly for Zn. We will focus on Scenarios 3 and 4.

Given the long retention time in the intermediate layer it may take centuries for the asymptotic concentrations to be achieved, and the time evolution of concentration must be considered. Furthermore, the maximum concentration in the upper layer may be greater than the asymptotic concentration, and may occur after a relatively short period (Fig. 10). Under Scenarios 3 and 4 the maximum upper layer concentrations for both Cu and Zn occur after 7–10 years. The maximum concentrations in Scenario 4 are very much greater than the asymptotic concentrations. Most interestingly, and perhaps most importantly, Scenario 4 results in a higher Cu concentration in the upper layer than Scenario 3, even though the NIS is directed into the intermediate layer and the removal rate increased to match the increased load on the intermediate layer. The result can be explained by noting that, for the Island Copper Mine pit lake, the retention time in the intermediate layer is almost 2 orders of magnitude greater than in the upper layer. Therefore, we can approximate Eq. (8*a*) by

$$C_1 \approx C_{1\infty} + (C_{2i} - C_{2\infty})(Q_2/Q_1)e^{-\tau_2} \qquad (10)$$

Our primary interest is in the maximum upper layer concentration, $C_{1\,\max}$. From Eq. (10) there are two possibilities

$$\text{If } C_{2i} < C_{2\infty}, \quad \text{then } C_{1\,\max} = C_{1\infty} \qquad (11a)$$

$$\text{If } C_{2i} > C_{2\infty}, \quad \text{then } C_{1\,\max} \approx C_{1\infty} + (C_{2i} - C_{2\infty})(Q_2/Q_1) \qquad (11b)$$

In the second case $C_{1\,\max}$ occurs early in the evolution. For Scenarios (3) and (4) Eq. (11*b*) gives $\mathrm{Cu}_{1\,\max} \approx 0.026$ and 0.039 mg/L, and $\mathrm{Zn}_{1\,\max} \approx 0.97$ and 0.59 mg/L, respectively. These values are all greater than the values computed using Eq. (8*a*), but never more than 5% greater, see Table 2. Therefore, we can use Eq. (11) to get a better understanding of the time evolution of concentration. If $C_{2i} < C_{2\infty}$, then Eq. (11*b*) can be rewritten as

Scenario 3

$$C_{1\,\max} \approx (C_N Q_N + C_{2i} Q_S + C_R Q_R - R_1)/Q_1 \qquad (12a)$$

Scenario 4

$$C_{1\,\max} \approx (C_{2i} Q_N + C_{2i} Q_S + C_R Q_R - R_1)/Q_1 \qquad (12b)$$

Unless otherwise stated we will assume $R_1=0$. Two interesting observations can be made using Eqs. (12) that could impact on future management strategies. First, if $C_{2i} > C_N$, then the maximum concentration is higher in Scenario 4 than Scenario 3, because the effect of diverting the NIS to the intermediate layer is to increase the flux of intermediate layer fluid into the upper layer. Second, when $C_{1\,\max}$ occurs early in the evolution it is essentially independent of R_2, because the peak occurs before the removal has a significant effect on the intermediate layer.

The calculations presented above are subject to considerable errors in the measurement of the input parameters (concentrations, flow rates, and removal rates). These parameters vary seasonally and from year to year, and are probably subject to long-term trends. Nevertheless, they may be useful in helping to formulate future management strategies. From Fig. 10 it seems that, if the pit lake were operating under either Scenario 3 or 4, then the permit levels for Cu and Zn would be satisfied. However, under Scenario 3, $Z_{1\,\max}=0.96$ mg/L, is only marginally less than the permit limit (1.0 mg/L) and could for any number of reasons exceed it. Diverting the NIS to the intermediate layer (Scenario 4) results in $Z_{1\,\max}=0.57$ mg/L, which offers a respectable factor of safety. On the other hand diverting the NIS to the intermediate layer increases $\mathrm{Cu}_{1\,\max}$ from 0.025 to 0.037 mg/L, approaching the permit limit of 0.05 mg/L. Diverting the NIS would also raise the DO input into the intermediate making it more difficult to achieve metal removal by SRB. Furthermore, the injection of NIS into the upper layer prevents upwelling—although, as mentioned above, only a small portion of the NIS discharge may be needed to maintain an upper layer of the desired thickness.

There are other issues to be considered. For instance, in Scenario 3 we have assumed that $R_1=0$ and $R_2=M_S$. Even if fertilization were discontinued there is the possibility of some removal from the upper layer, which from Eq. (12*a*) would help. However, we have already noted that a higher R_2, even if achievable, would not help in the short term. One strategy would be to wait for a reduction in C_{2i} before halting fertilization, however from Eq. (12*a*) and Table 1 it is apparent that $C_{1\,\max}$ is not particularly sensitive to changes in C_{2i}. Finally, there are other metals subject to permit limits (e.g., cadmium) that may need to be considered, see Poling et al. (2003).

If reliable and economic long-term management strategies are to be formulated current monitoring efforts need to be continued, and efforts made to better understand biogeochemical removal processes and their effectiveness. The above equations could be used to suggest if, and when, fertilization could be halted with the confidence that natural biogeochemical removal processes would ensure compliance with permit limits. Fortunately, there is scope for experimentation since the relevant time scales involved are long enough that, if upper layer metal concentrations approached unacceptable levels, then fertilization could be restarted.

Conclusions

The Island Copper pit lake has evolved into three distinct layers: a brackish and well-mixed upper layer; a plume stirred intermediate layer; and a thermally convecting lower layer. The initial closure plan proposed the establishment of a “passive” treatment system removing heavy metals by metal-sulfide precipitation via anaerobic SRB, once anoxic conditions were established in the intermediate and lower layers. Anoxia has been achieved in the lower layer, but oxygen consumption rates have been less than initially predicted and anoxia has yet to be achieved in the intermediate layer. Without metal-sulfide precipitation (or any other form of biogeochemical removal) in the intermediate layer, metal concentrations in the upper layer of the pit lake are likely to increase beyond the permit limits.

Consequently, two modifications have been made to the operation of the pit lake as a treatment system for ARD: fertilization of the surface waters, and the diversion of the north injection system (NIS) to the upper layer. Fertilization of the surface waters stimulates plankton blooms that absorb contaminant heavy metals, die, and sink to the sediments. The increase in organic matter in the intermediate and lower layers also boosts the oxygen demand in these layers. Diversion of the NIS to the upper layer provides a useful management option to reduce the oxygen input to the intermediate layer, to slow down the dilution of the intermediate layer, and to help maintain an upper layer that is sufficiently thick to resist wind-driven upwelling.

Fertilization of the surface waters has been successful in treating large quantities of ARD and producing outflowing water well below effluent permit levels for all dissolved metals (Poling et al. 2003). Predictions of possible long-term trends for Cu and Zn concentrations have been obtained by assuming that the intermediate and upper layers behave like continuously stirred reactors of constant volume. When the intermediate layer becomes anoxic, biogeochemical removal rates from the intermediate layer may be high enough to ensure that Cu and Zn concentrations in the upper layer remain below the permit levels, even if there is no biogeochemical removal from the upper layer. If this is the case fertilization will no longer be essential, and a passive treatment system will be possible. Continued monitoring of the pit lake is required so that biogeochemical removal rates can be calculated and used to help make long-term predictions and management decisions.

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Notation

The followings symbols are used in this paper:

C = concentration;
M = mass flux of dissolved metal;
P = pressure;
Q = volumetric flow rate;
R = dissolved metal removal rate;
S = salinity;
T = temperature;
t_r = retention time;
V = volume;
ρ = density; and
τ = t/t_R = dimensionless time.

Subscripts

A = ARD;
i = initial conditions;
j = layer number;
N = north injection system;
P = precipitation;
R = runoff;
S = south injection system;
s = sediments;
1 = upper layer;
2 = intermediate layer; and
3 = lower layer.

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