Water Quality Effects from an Aquaculture Operation in a Meromictic Iron Pit Lake in Northwestern Ontario, Canada

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In 1989, aquaculture for the production of rainbow trout (*Oncorhynchus mykiss*) was started in the upper depths of the 180-m deep Caland pit lake formed after the flooding of the former Steeprock Iron Mine. After ten years of operation, water quality was shown to be affected by the fish farm as well as proximal waste and the surficial geology of the area. Meromictic conditions occurred in the pit lake with a dichotomy in water quality versus depth. A cluster of parameters (ammonia, total phosphorus, total Kjeldahl nitrogen, dissolved organic carbon, B, pH, dissolved oxygen) were either in higher concentrations in the mixolimnion or varied little throughout the entire water column, and a second cluster of parameters had higher levels below the chemocline (conductivity, alkalinity, dissolved inorganic carbon, total dissolved solids, nitrate, sulfate, Ca, Mg, hardness, Na, Cl, K, Fe, Mn, Sr, and silicon dioxide). Statistically significant variations in concentrations among many of these chemical parameters were observed between the two years of study, among depths, and among seasons. Nutrient loading from excess fish food and waste increased phosphorus and ammonia levels. The expansion of the fish farm also reduced the levels of dissolved oxygen in the lower depths of the pit lake, thus reducing the volume of usable water for aquaculture purposes. In the early 2000s the fish farm decreased production and those closed in 2006. Later data showed that nitrate and potassium had corresponding decreases in both the monimolimnion and the mixolimnion. Sulfate levels remained elevated.

Key words: pit lake, aquaculture, meromictic, northwestern Ontario

Introduction

The closure of open pit mines and their subsequent filling with both ground and surface waters has produced many pit lakes (Miller et al. 1996; Banks et al. 1997; Doyle and Runnells 1997). These pit lakes differ from natural lakes. They are usually very deep, have low surface to depth ratios, often lack any outflow to the surrounding watershed, and in addition to thermal stratification, often have chemical stratification caused by runoff from the newly scoured surfaces of the open pit operations (Hamblin et al. 1997).

It has been suggested that if there is sufficient acid-buffering capacity, these closed lake systems could be used for aquaculture as an alternative approach to conventional mine closure rehabilitation. In northern Minnesota and in northern Quebec, aquaculture operations were established in several abandoned open pits that had moderately hard water (Cornel and Whoriskey, 1993; Axler et al. 1996). Unfortunately the development of anoxic conditions in the hypolimnion, as chemical and microbial oxidation of fish food and feces occurs, may result in eutrophic conditions and adversely affect the water quality of the receiving water body (Axler et al. 1992). Nevertheless, the premise of utilizing an abandoned mine site for aquaculture remains attractive from both an economic and social sense.

If such pit lakes are to be used for aquaculture, it is necessary to identify the potential impact of these operations on the environment and develop techniques that can minimize or remediate any problems. The first step in this process, and the objective of this study, is to separate the impacts of the aquaculture operation on water quality from those occurring from the weathering of exposed mine rock and tailings.

Methods

Study Site

The study site consists of flooded open pits located in the former middle and east arms of Steep Rock Lake (48°48'N, 91°39'W) near Atikokan, Ontario, Canada (Fig. 1). The largest pits are Hogarth in the middle arm and Caland pit lake in the east arm. These sections of Steeprock Lake were isolated from the main lake in the early 1940s by a series of diversions and dams. The area was drained, its overburden removed, and the ore extracted. Mining occurred from 1944 to 1979. After mining ceased, the pits began to fill with water from rainfall, runoff, and groundwater seepage from their drainage basin which encompasses an area of 16.8 km². As the basin fills, the pits will eventually join and reach depths greater than 300 m before discharging into the west arm of Steeprock Lake and from there into the Seine River.

The Steep Rock area contains Archean metavolcanic, metasedimentary, and intrusive rocks which have been displaced by a series of faults (Shklanka 1972). These rocks lie on weathered granite rocks which separate the main sections of the ore body.

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The ore is bordered by "paint rock," a soft clay-like material eroded from the limestone, and by "ash rock" formed from volcanic ash. A footwall carbonate formation, which varies in composition from calcite ($CaCO_3$) to dolomite ($CaCO_3 \cdot MgCO_3$), underlies the paint rock. There are also small patches of conglomerates made up of Archean sand and gravel deposited by an ancient sea (Pye 1968). The displacement of the granitic masses along the faults caused folding and tilted the ore body and associated rocks to a near vertical position. The ore extended to a depth of 760 m, width of 50 m, and length of over 4 km. The ore consisted mostly of goethite and hematite and averaged 56.5% Fe, 3.42% Si, 0.17% P, 8%Al, and 0.21% Mn (Steep Rock Mines 1943). Although it is possible the iron originated from volcanic exhalations, the most accepted theory for iron deposition in the region is that the iron was precipitated as oxides in the shallow waters of a Precambrian sea in limited areas where oxygen became abundant in the water column (Cloud 1973; Ojakangas and Matsch 1982). The occurrence of large fossilized blue-green algal mounds at the Hogarth pit gives credence to this hypothesis.

The present study is concerned with Caland pit lake. By 1999, the depth and area of Caland pit lake were approximately 180 m and 120 ha, respectively. The pit lake is steep sided and well sheltered from winds.

Snow Lake Fish Farm commenced cage culture with chinook salmon (*Oncorhynchus tshawytscha*) in 1989 in Caland pit lake. By 1992 salmon production reached 91,000 kg total weight, but was discontinued since food conversion was inefficient. In 1993 rainbow trout (*Oncorhynchus mykiss*) production expanded, reaching 57,000 kg by 1995. In 1996, because of damage to the farm from ice and storms, rainbow trout production decreased to 36,000 kg. In 1997 production further declined to 20,000 kg. In 1998 a major expansion of the facility was undertaken, and by the end of the year there were 450,453 fish with a biomass of 84,629 kg. In December 1999 fish numbers averaged 733,325 with a biomass 163,325 kg, and by August 2000 production reached 227,000 kg. With increases in fish, the amount of fish food utilized also increased from 146,000 kg in 1998 to 214,000 kg in 1999.

Field Procedures

Figure 1 shows the locations of the three sampling stations in the pit lake and the four drainage sampling stations. The three sampling stations were the same as those used by the Ontario Ministry of the Environment in 1992.

In the first year, sampling was conducted monthly from June to September, 1998, before ice cover in early November and after ice cover in late February 1999, for a total of six times. In the second year, the same sampling regime was followed except it started immediately after ice-out in mid May 1999, for a total of seven sampling times.
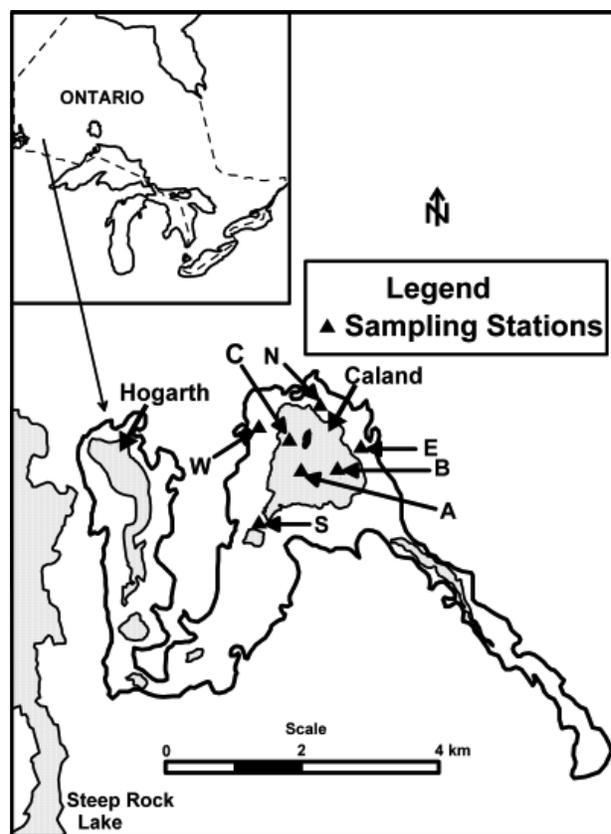

Fig. 1. Steeprock mine site consisting of Caland pit lake and Hogarth pit lake. The black outline indicates the eventual area that will be flooded and overflow into the Seine River. Sampling occurred at three water chemistry stations in Caland pit lake. Station A was beside the fish cages, 180m deep, B was 90m deep, and C was 120m deep. Four drainages (north [N], east [E], south [S], and west [W]) were also sampled.

At the three sampling stations in Caland pit lake, water was collected from the mixolimnion (1-m depth), chemocline, monimolimnion (30 m), and 1 m above the bottom of the pit. Dissolved oxygen (DO) and temperature profiles were obtained from each station using a Yellow Springs Instrument (YSI) Model #51B. Drainage water was collected by surface grab method using polyethylene bottles. In addition to the routine samples, field sampling quality controls were collected each sampling period, and included one field blank, one travel blank, and one duplicate sample.

Analytical Procedures

All in-lab analyses adhered to strict quality assurance/quality control (QA/QC) protocols. A blank sample was run at the beginning of each tested parameter, followed by a standardized QA/QC sample and a repeat of the succeeding field sample. The QA/QC and the repeat samples were tested after every ten field samples. Each of the tests followed Lakehead University Environmental Lab Standard Operating Procedures (LUEL 2000)

which were modified from *Standard Methods for the Examination of Water and Wastewater* (APHA 1992).

Alkalinity, pH, and conductivity were measured within 24 hours of sampling after samples reached ambient laboratory temperature. A 50-mL aliquot of the unpreserved water sample was analysed for alkalinity and pH using the automated Mettler DL20 Compact Titrator with Mettler Probe #DG115-SC. Conductivity was measured with a VWR Digital Conductivity Meter with automated temperature compensation calibrated at 200 μS·cm^{-1}.

Total suspended solids (TSS) and total dissolved solids (TDS) were determined gravimetrically and reported as weight per volume of sample. TSS refers to all material (residue, particulate) that was removed by filtration through a 0.45-μm glass fibre filter and dried at 103°C. The TDS fraction measured the amount of dissolved material that passed through a 0.45-μm filter. This material was collected in beakers and dried at 103°C. The filters and beakers were desiccated for 12 hours, and then final weights were obtained to five decimal places. An aliquot of the unused portion of the filtered TDS sample was used to determine anion (sulfate [SO_4], Cl, and nitrate [NO_3]) concentrations using a Dionex DX-120 Ion Chromatograph (IC) in conjunction with an AS40 automated sampler.

Dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC) were measured by converting any carbonate and bicarbonate species to carbon dioxide. DIC was measured by acidifying the sample and passing it over a carbon dioxide permeable membrane. The CO_2 gas passed through the membrane and was absorbed in a weakly buffered alkaline solution containing a phenolphthalein indicator. The decrease was detected by the Skalar San^{PLUS} System. DOC was measured when the sample was introduced into the Skalar where it was acidified and flushed with nitrogen gas to remove the inorganic carbon. The sample was then oxidized with an ultraviolet (UV) digester in an acid-persulfate medium. Again, the carbon was converted to CO_2, and following dialysis was measured by determining the loss in absorbance of a weakly buffered alkaline solution containing a phenolphthalein indicator.

Reactive silica (SiO_2) was analysed using the molybdate-reactive silica method. This method was based on the formation of the molybdenum heteropoly blue complex and measures only dissolved reactive silicate anions (APHA 1992). Silica in the soluble colloidal form or in the insoluble polymerized form does not react and thus was not detected by this method. Ammonium molybdate at pH 1.2 reacted with silicates to produce a yellow molybdosilicic acid complex. Since phosphates also react in this manner, oxalic acid was used. The Skalar was used to measure the absorbance of the coloured solution at 660 nm. The absorbance was proportional to the reactive silicate concentration in the original sample.

Total phosphorus (TP) and total Kjeldahl nitrogen (TKN) were digested then analyzed using a colourimetric determination on the Skalar Autoanalyser system. Before analysis, samples were treated with sulfuric acid and digested stepwise to 400°C to achieve a three-fold preconcentration. The samples were then restored to their original volume. TKN concentrations were determined spectrophotometrically at 660 nm based upon a modified Berthelot (APHA 1992) reaction. TP was simultaneously analyzed on the same sample aliquot. The digestion converted all forms of phosphorus to orthophosphate which was determined colourimetrically at 880 nm based on the ascorbic acid procedure (APHA 1992).

An aliquot of the sulfuric acid-preserved water was used for the determination of ammonia nitrogen (NH_4-N). The method was also based on the modified Berthelot reaction: ammonia was chlorinated to monochloramine which reacted with salicylate to 5-aminosalicylate. After oxidation and oxidative coupling, a green-coloured complex was formed. The absorption of the formed complex was measured spectrophotometrically at 660 nm using the Skalar.

Total metals analysis was performed after a nitric acid digest at 100°C for 12 to 24 hours resulting in a two-fold concentration of the sample. Samples were analysed using the Jarrell Ash Inductively Coupled Argon Plasma 9000 Spectrometer (ICP) for Fe, Mn, Zn, Cu, Ni, Al, B, Co, Cr, Sr, Ca, K, Mg, and Na. Only those elements that were above detection limits were included in the results.

Total hardness, expressed as calcium carbonate in milligrams per litre, is defined as the sum of the calcium and magnesium concentrations. The preferred method to compute hardness was taken from the results of separate ion determinations of calcium and magnesium in the calculation: hardness (mg·L^{-1} of CaCO3) = 2.497 × (Ca [mg·L^{-1}]) + 4.118 × (Mg [mg·L^{-1}]) (APHA 1992).

Sulfide was analyzed using a Cary SE UV-Vis-NIR Varian Spectrophotometer, based on a method outlined by the United States Environmental Protection Agency (1988).

Data Analysis

In some cases values for the chemical parameters were less than the mean detection limit (MDL), and this led to a problem with some of the statistical procedures. Such data were handled by a) removing the parameters from the analyses if the majority of the data (>90% in each parameter) were less than the MDL and b) for those parameters that had only a few values less than the MDL, the value was replaced with the MDL/2. Data analysis then proceeded in three steps:

(i) A hierarchal cluster analysis program (SPSS 2001) grouped the chemical associations in the water column in 1998 and 1999.

(ii) In order to determine the short term effects of the fish farm, repeated measures ANOVAs (analysis of variances) (SPSS 2001) determined whether the water chemistry parameters for the 1998 and 1999 data differed in concentrations between depths

(mixomolimnion, chemocline, monimolimnion) and/or years.

(iii) Longer term effects of the fish farm were assessed by comparing data from June 1992, with June 1998 and 1999, for depths of 1 m (mixolimnion), 18 m (chemocline), and 30 m (monimolimnion). The 1-m-off-of-the-bottom data was not available for 1992. Repeated measures ANOVAs were again used for this procedure.

Results

Meromictic Conditions

The results (Table 1, Fig. 2) suggested that Caland pit lake could be considered meromictic with water quality in a mixolimnion (epilimnion) that was distinctly different from a monimolimnion (hypolimnion). The chemocline started at 23 m and continued for approximately 5 m. Many of the water chemistry parameters had depth distributions typical of a meromictic lake with elevated concentrations below the chemocline. Hydrogen sulfide was less than 0.1 mg·L⁻¹. The clinograde oxygen profiles were also indicative of a meromictic environment with complete oxygen depletion below the mixolimnion (Fig. 2). With turnover, the oxygen concentrations replenished in the mixolimnion to a depth of 25 m. This is shown by the oxygen profile in the 2000 winter data (Fig. 2). The 1998 and 1999 August temperature profiles showed thermal stratification in the mixolimnion with a defined thermocline and a hypolimnion that commenced at 15 m (Fig. 2). During the winter, ice did not form above the fish pens due to the constant fish movement, but was greater than 0.5-m thick elsewhere on the lake.

Cluster analysis of 1998 and 1999 data (Fig. 3) demonstrated this dichotomy in water quality. In both years there was a cluster of parameters that had higher levels below the chemocline (Cluster 1: conductivity, alkalinity, DIC, TDS, NO_3, SO_4, Ca, Mg, hardness, Na, Cl, K, Fe, Mn, Sr, and SiO_2) and a cluster or parameters (Cluster 2: NH_4, TP, TKN, DOC, B, pH, Temperature, DO) that were either in higher concentrations in the mixolimnion or varied little throughout the entire water column. In 1999, TP was included in the first cluster as a result of increased levels off-bottom. ANOVAs revealed that all parameters except Fe varied significantly with depth, and all parameters varied significantly with season with the exception of Fe, TKN, Na, K, TDS, and B (Table 1). The onset of winter and the corresponding turnover in the shallower regions of Caland pit lake was the main cause for variations in many of the parameters in the second cluster. DOC showed several peaks throughout the sampling season, which coincided with rainfalls.

TABLE 1. Caland pit lake water chemistry mean values [a]

	Mixolimnion		Chemocline		Monimolimnion		Off-bottom	
	1998	*1999*	*1998*	*1999*	*1998*	*1999*	*1998*	*1999*
Alkalinity [b,c,d]	125	124	129	125	167	164	172	173
Conductivity [b,c,d]	702	604	718	620	1,056	890	1,131	1,008
DIC [b,c,d]	32.3	27	33.7	28.7	43.7	39.6	46.1	42.3
DOC [b,c,d]	3.7	3.5	3.7	3.4	3.4	3.2	3.4	3
Hardness [b,c,d]	360	334	370	343	593	544	644	629
Cl⁻ [b,c,d]	7.5	6.3	7.6	6.3	12.9	10.4	13.5	11.5
NO_3^- [b,c,d]	2.44	1.92	2.58	1.98	4.4	3.21	4.2	2.83
SO_4^- [b,c,d]	243	217	243	223	443	348	497	417
Ca [b,c,d]	84	78	87	81	130	121	140	136
K [b,d]	4.3	3.9	4.3	3.9	5.2	4.8	5.8	5.1
Mg [b,c,d]	37	34	37	35	65	59	72	70
Na [b,d]	12	11.1	12.2	11.4	18.2	16.8	19	18.7
B [b,d]	0.06	0.13	0.07	0.16	0.07	0.16	0.07	0.15
Fe	0.02	0.02	0.02	0.02	0.03	0.02	0.04	0.05
Mn [b,c,d]	0.01	0.01	0.01	0.01	0.07	0.07	0.18	0.33
Sr [c,d]	0.62	0.6	0.63	0.62	0.97	0.92	1.02	1.05
NH_4 [b,c,d]	0.05	0.14	0.06	0.17	0.03	0.06	0.03	0.07
pH [c,d]	7.9	7.76	7.58	7.49	7	7.13	6.98	7.09
SiO_2 [b,c,d]	2.57	2.91	3.46	3.11	9.78	8.31	10.8	9.07
TKN [d]	0.23	0.38	0.26	0.45	0.29	0.34	0.25	0.32
Total - P [b,c,d]	0.009	0.011	0.008	0.014	0.006	0.01	0.006	0.02
TDS [b,d]	479	450	491	459	770	707	859	838
DO [b,c,d]	9.26	9.69	9.54	8.18	0.55	0.38	0.3	0.27

[a] Units are mg·L⁻¹, except conductivity (µS·cm⁻¹) and pH.
[b] Significant ($p < 0.05$) difference between years.
[c] Significant ($p < 0.05$) difference between seasons.
[d] Significant ($p < 0.05$) difference between depth.
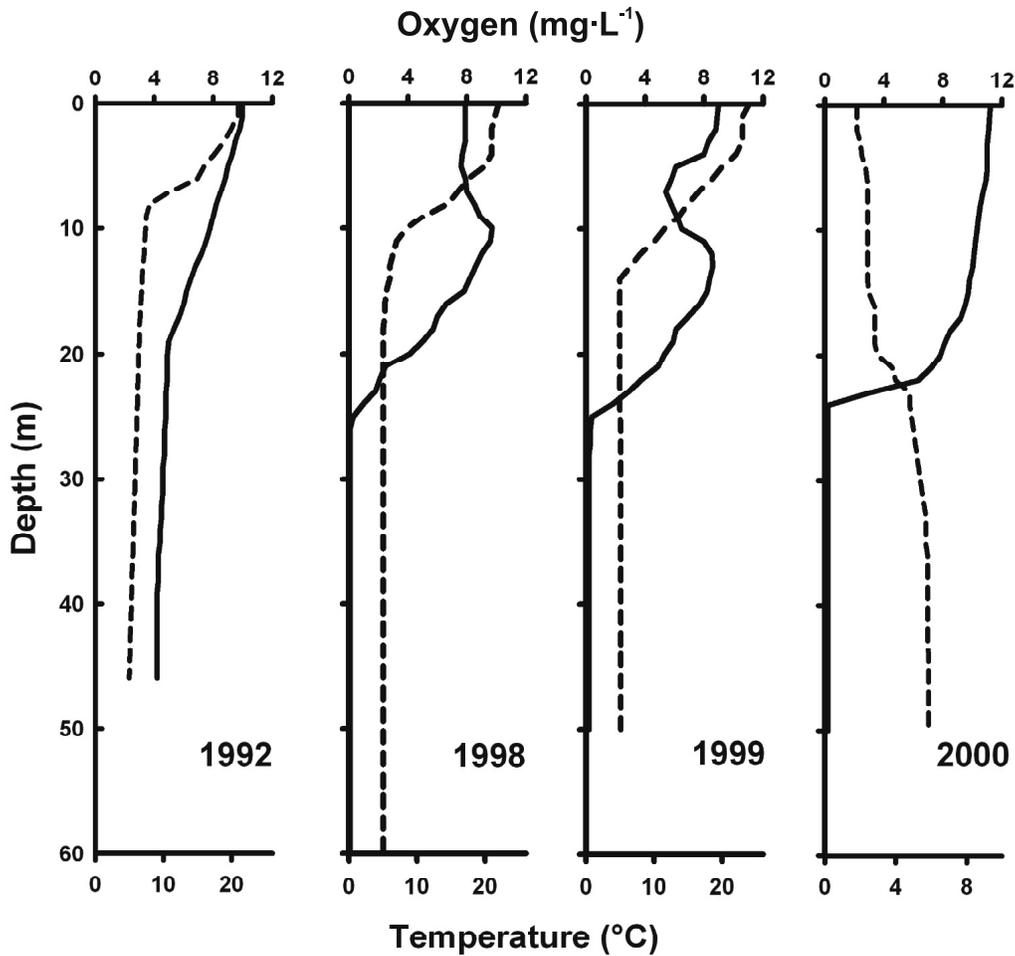
Oxygen (mg·L^{-1})

Fig. 2. Mean values for dissolved oxygen (—) and temperature (---) profiles for June 1992 (MOE unpublished data), August 1998 and 1999, and winter 2000. The profiles are averages of the three sampling stations in Caland pit lake, Ontario.

Water Quality Variations Between 1998 and 1999

Repeated measures ANOVAs determined which variables were significantly different ($p < 0.05$) between 1998 and 1999 (Table 1). In the monimolimnion (Cluster 1) all variables decreased in concentration from 1998 to 1999 except Fe, Sr, and Mn, which remained relatively constant. In the mixolimnion, on the other hand, NH_4, TP, and B increased while DOC and DO decreased. The variables which decreased in concentration in the mixolimnion and monimolimnion likely reflect dilution from increased rainfall in 1999 (889.1 mm) versus 1998 (598.4 mm), and lower temperatures (mean temperature May to October in 1999 was 13.7°C versus 14.7°C in 1998). Higher rainfall in 1999 also tended to dilute concentrations of measured parameters in the inflow drainage (Table 2). Changes in the mixolimnion, however, reflect expanded production of the fish farm, causing both NH_4 and TP to increase and DO to decrease.

Water Quality Changes Since 1992

Repeated measures ANOVAs determined that there were significant differences from 1992 to 1998 to 1999 for conductivity, alkalinity, Ca, TP, TKN, SO_4, Mg, hardness, Cl, NO_3, K, Na, pH, and TDS . Figure 4 shows some representative parameters. Variables that comprised Cluster 1 (Fig. 3), for example, alkalinity, conductivity, calcium and sulfate, declined in concentration in the mixolimnion and chemocline while changing little in the monimolimnion. On the other hand, the mixolimnion variables (Cluster 2), fluctuated considerably with depth from 1992 to 1999. For example, P declined from 1992 to 1998 in the chemocline and monimolimnion, and then increased in these two layers in 1999. TKN decreased significantly in concentration in the monimolimnion and chemocline from 1992 to 1998.

The most noticeable change was the depletion of oxygen in the monimolimnion. In July 1992, DO levels
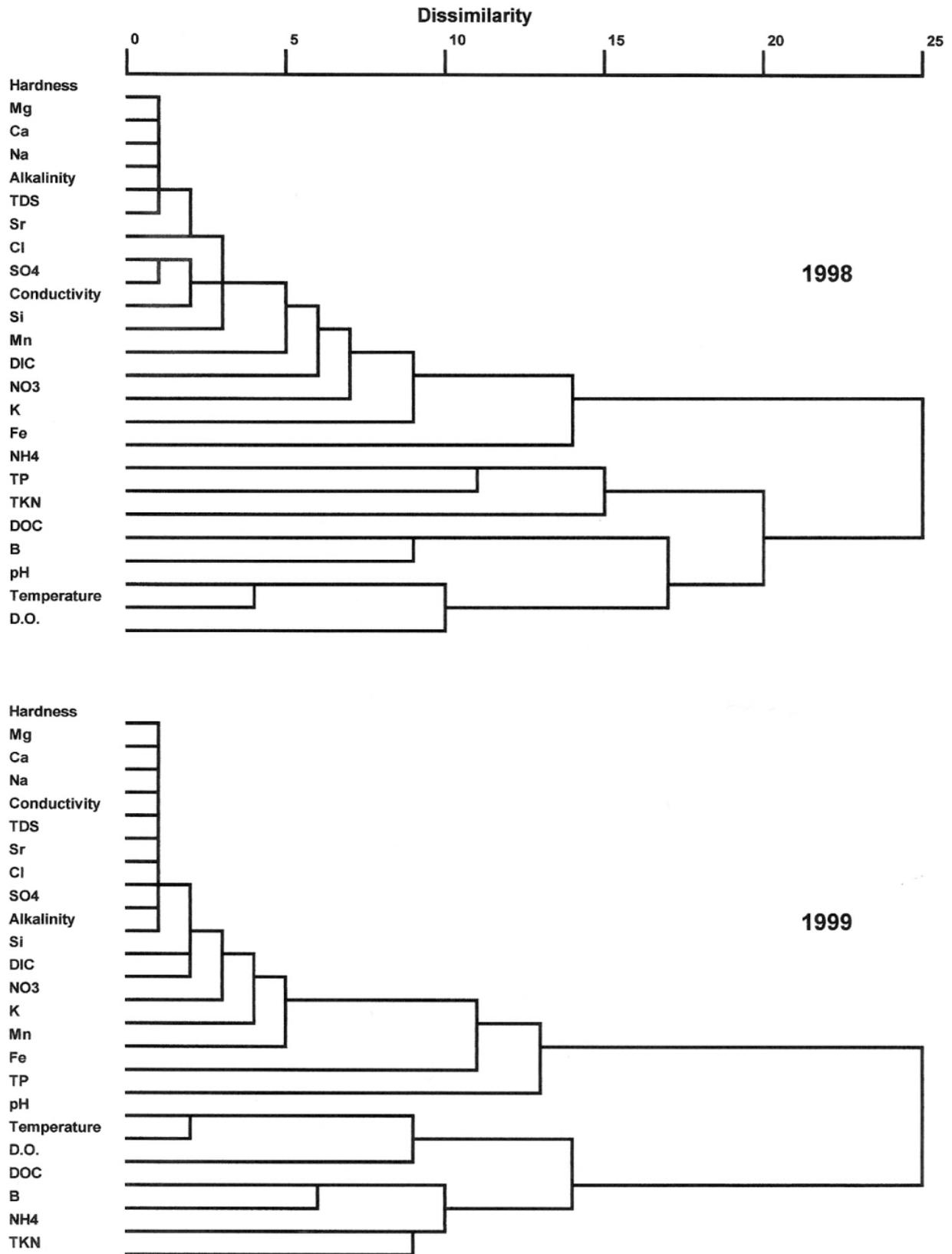
Fig. 3. Hierarchal cluster analysis using the Pearson correlation average linkage between groups for the water chemistry parameters measured in 1998 and 1999 in Caland pit lake, Ontario.

TABLE 2. The 1998 and 1999 inflow drainage water chemistry mean values for Caland pit lake[a]

	North		South		West		East	
	1998	*1999*	*1998*	*1999*	*1998*	*1999*	*1998*	*1999*
Alkalinity	119	124	122	139	36	40	117	115
Conductivity	263	241	1,639	1,134	1,728	499	278	240
DIC	27.6	27.6	34.2	25.2	9.7	9.1	29.5	25.9
DOC	4.6	4.4	5.9	7.1	3.2	3.5	4.2	4.5
Hardness	133	134	1,049	742	350	278	130	122
Cl^-	3.3	5.5	3.2	2	3.1	0.6	7.6	6.4
NO_3^-	0.03	0.05	0.01	0.01	0.01	0.01	0.11	0.08
SO_4^-	11.7	43.7	867.2	573.2	851.1	237.6	28.1	19.1
Ca	41.07	41.79	163.47	130.3	93.69	75.98	37.56	35.28
K	2.5	1.7	4.6	3.9	2.4	2	1	1.9
Mg	7.42	7.24	155.57	101.39	28.31	21.5	9	8.28
Na	4.28	4.01	12.43	8.83	2.1	1.88	7.95	7.39
Al	0.17	0.26	0.09	0.19	<0.05	0.07	0.06	0.06
B	0.07	0.15	0.22	0.12	0.08	0.15	0.1	0.15
Fe	0.25	0.26	0.06	0.09	0.07	0.06	0.07	0.08
Mn	0.02	0.03	0.13	0.31	0.24	0.91	0.01	0.01
Sr	0.111	0.111	0.63	0.43	0.231	0.2	0.139	0.138
pH	8.11	8.19	8.05	8.12	6.97	6.84	7.51	7.83
SiO_2	12.04	7.42	1.24	3.82	6.04	4.91	5.98	3.58
TKN	0.14	0.12	0.24	0.33	0.25	0.18	0.12	0.16
TDS	154	153	1,420	1,002	508	554	192	145

[a] Units are $mg \cdot L^{-1}$, except conductivity ($\mu S \cdot cm^{-1}$) and pH.

of 4 $mg \cdot L^{-1}$ extended at least to a depth of 45 m, while in 1998 and 1999 DO concentrations were less than 1 $mg \cdot L^{-1}$ below a depth of 23 m.

Discussion

Development of Meromixis

The results (Fig. 2) showed that Caland pit lake has become meromictic with a defined chemocline. Meromixis commonly occurs from either crenogenic causes due to the influx of saline groundwater into freshwater lakes and conversely surficial freshwater into a saline lake (Jellison and Melack 1993), or from biogenic mechanisms when there is an accumulation of solutes in the water column derived from the decomposition of organic matter (Walker and Likens 1975). In the case of Caland pit lake , surface water drainage was the main cause. Drainage from the southern and western sections of the basin generally had elevated levels of the major components of the monimolimnion (Table 2). The western drainage was the only area where the water was less than a pH of 7, and the reduced alkalinity from this drainage indicates that acid formation was occurring in this area. Apparently, the presence of limestone and dolomite (McNaughton et al. 1999) are able to buffer such acidic influxes. Drainage from the north and east was much more dilute (Table 2). This was caused either by variations in geology in this section of the basin or more channelized flow that was less impacted by contact with the exposed surface. Drainage from these areas was the source of the mixolimnion. Continued flow from these dilute drainages tended to lower the concentrations of the major cations and anions from 1992 to 1999 (Fig.

4) at all depths. The net result was a meromictic pit lake that could be characterized as a hard water, alkaline lake with a relatively high salt concentration indicated by its specific electrical conductivity and TDS values averaging 650 $\mu S \cdot cm^{-1}$ and 470 $mg \cdot L^{-1}$ in the mixolimnion to 1,090 $\mu S \cdot cm^{-1}$ and 760 $mg \cdot L^{-1}$ in the monimolimnion, respectively (Table 1). Cations were dominated by Ca/Mg and the major anion was SO_4. The concentration gradients for variables associated with the monimolimnion (Fig. 3) versus the mixolimnion have increased since 1992 (Fig. 5), indicating that the two layers are becoming more distinct for many chemical parameters.

The occurrence of SO_4 as the major anion is in contrast to the findings of Axler et al. (1992) who determined that pit lakes in northern Minnesota were dominated by HCO_3^-. Other more saline meromictic lakes like Big Soda Lake in Nevada (Kimmel et al. 1978) and Mono Lake in California (Jellison and Melack 1993) commonly have very high concentrations of SO_4. Sulfur-reducing bacteria convert the SO_4 to H_2S which can reach concentrations of over 400 $mg \cdot L^{-1}$ (Cloern et al. 1983). This is not occurring in Caland pit lake as sulfide concentrations were less than 0.1 $mg \cdot L^{-1}$.

Essentially, a chemical gradient in Caland pit lake was formed that completely isolated any changes in the monimolimnion from the mixolimnion. The reverse, however, was not true. The fish farm dramatically altered the water quality throughout the water column.

Influence of the Snow Lake Fish Farm

Between 1998 and 1999 there was an eight-fold increase in fish production (from 20,000 kg early in 1998 to

163,325 kg by the end of 1999). Associated with this increase In fish production was a compounding problem of waste production, both with food and feces. Bergheim et al. (1991) stated that only 25 to 30% of the nitrogen and phosphorus in feed is typically retained in the fish, with the remainder being lost to the environment. Undigested feed is excreted in the feces as solid waste, and the by-products of metabolism (ammonia, urea, phosphate, carbon dioxide, etc.) are excreted as dissolved wastes mostly by the gills and kidneys (Ackefors et al. 1994; Bureau and Cho 1999). Nitrogen excretion resulting from dietary protein oxidation is the major source of dissolved nitrogenous wastes. High rates of nitrogen loading as excreted ammonium may contribute

to excessive algal growth, increased oxygen depletion (via nitrogenous biochemical oxygen demand), and potentially to un-ionized ammonia (NH_3) toxicity (Axler et al. 1996). In the specific case of Snow Lake Fish Farm, we estimated that approximately one tonne per year of organic matter accumulates in the bottom of the nets, which eventually is released to the bottom strata. These effects influenced various cations, dissolved oxygen, phosphorus, and nitrogen.

Cation concentrations were lower in the mixolimnion in 1999 versus 1998. Although dilution was likely the main cause of the lowered concentrations, complexing and/or sorption on substrates such as clays, organic
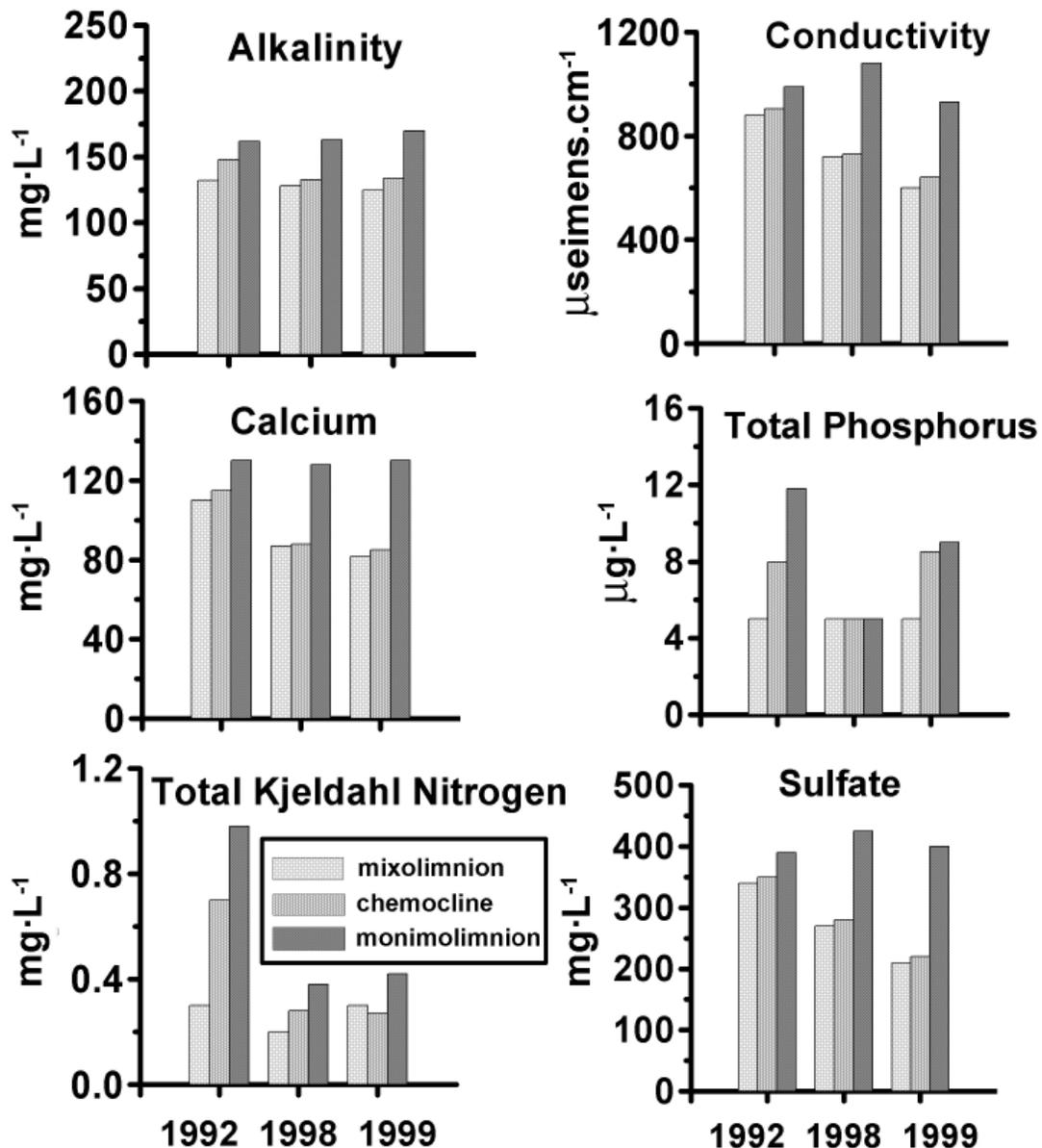

Fig. 4. A comparison of mean values for representative water chemistry parameters collected in June 1992, 1998, and 1999 in Caland pit lake, Ontario. The 1992 data were collected by MOE (unpublished data). Represented depths are: mixolimnion, chemocline, and monimolimnion.
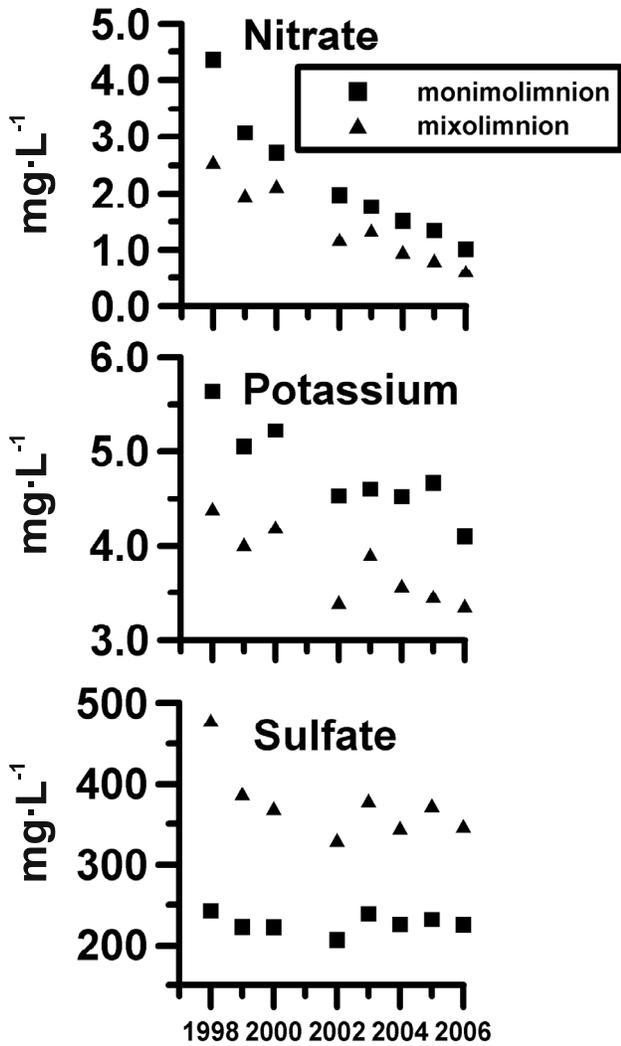
Fig. 5. Changes in nitrate, potassium and sulfate levels in the mixolimnion and monimolimnion of Caland pit lake from 1998 to 2006. Later data are from VanCook (2005) and Goold (2008).

matter, and aluminum, manganese, and iron-oxides are also known to reduce cation levels in pit lakes (Miller et al. 1996). Increased production from the fish farm operation with a corresponding increase of waste particles in the water column therefore may have further lowered the concentrations of these positively charged parameters.

The dramatic decline in oxygen levels since 1992 (in July 1992 greater than 4 mg·L^{-1} of DO was present at 45 m, but by 1998 concentrations were less than 1 mg·L^{-1} below a depth of 23 m [Fig. 3]) must be attributed to the fish farm activities which increased the biochemical oxygen demand of the mixolimnion. The development of similar anoxic conditions has been reported in the hypolimnion of other fish farm operations (Axler et al. 1992, 1996; Gale 1999). Between 1998 and 1999, the DO profile was further degraded as it developed a

distinct negative heterograde shape in the mixolimnion (Fig. 3). During the winter, ice did not form above the fish cages. A combination of these openings and the continuous movement (literally resembling egg beaters) of the caged fish ensured dissolved oxygen replenishment in the mixolimnion to a depth of 25 m (Fig. 3). This replenishment may not have occurred if ice covered the entire pit lake.

A second mechanism that can reduce oxygen levels in meromictic lakes is the oxidation of any H$_2$S produced by sulphur-reducing bacteria. The highest concentrations of these bacteria are found immediately below the chemocline and above the sediment where there is an abundance of organic matter. Should overturn of the mixolimnion (or the entire water column) occur, H$_2$S is oxidized spontaneously by any oxygen present with a corresponding lowering of DO (Jorgensen et al. 1979). However, as mentioned previously, sulfide levels are very low in Caland pit lake and so this would not be a factor in this instance.

The expansion of Snow Lake Fish Farm rainbow trout production from 1998 to 1999 coincided with an increase in TP concentrations. TP in 1998 was generally greater than the MDL (0.005 mg·L^{-1}), but increased at all depths in 1999, especially off-bottom (average 0.02 mg·L^{-1}) and at the thermocline (Fig. 3). Comparatively, in the Minnesota Aquafarms salmonid operation of approximately 1.8 million kg, Twin City-South net pen activities caused the phosphorus to increase from 0.01 mg·L^{-1} in 1988 to 0.094 mg·L^{-1} in 1992, and the Sherman pit TP levels rose from <0.01 mg·L^{-1} before aquaculture began to levels above 0.25 mg·L^{-1} towards the end of operation in 1995 (Axler et al. 1996). Ackefors and Enell (1990) determined that 80% of the phosphorus load from net pen aquaculture is associated with solids which settle quickly to the bottom. Increased hypolimnetic anoxia associated with the decomposition of these particles could have accentuated P release from the sediment causing the observed elevated levels at the sediment-water interface in Caland pit lake in 1999.

One encouraging result was the observed recovery of ambient P concentrations once fish farming activities declined. In 1992, TP was approximately 0.01 mg·L^{-1} which was associated with the density and feeding of chinook salmon (91,000 kg). Reduced production of fish from 1992 to 1998 caused a decline in TP to <0.005 mg·L^{-1} by early 1998 (Fig. 4). Similarly, following the closure of Minnesota's net pen activities, phosphorus levels decreased from 0.094 mg·L^{-1} in 1992 to near 0.010 mg·L^{-1} in 1994 (Axler et al. 1996).

In 1999, both TKN and NH$_4$ approximately doubled in concentration from 1998. This was consistent with the findings of Axler et al. (1996) who found in Minnesota that nitrogen levels (NO$_3$ + NO$_2$ + NH$_4$) rose from 0.363 mg·L^{-1} in 1988 to 2.043 mg·L^{-1} in 1993 in response to intensive net pen aquaculture. A particular problem is NH$_4$ and specifically the amount of toxic free ammonia that would be present. The amount of toxic

free ammonia increases with temperature and pH. For example, the percentage of un-ionized ammonia at a temperature of 15°C and pH of 7.5 is 0.859, whereas at pH 8.0, the percentage is 2.67 (Ackefors et al. 1994). Estimated levels of toxic free ammonia in Caland pit lake would still be well below concentrations that would be detrimental to fish, but if the observed trends continued, this could become a problem. In 1999, the NO_3-N levels decreased at each depth in Caland pit lake (Table 1), likely from denitrification caused by increasing anoxia in the monimolimnion. Similarly, Axler et al. (1995) suggested denitrification could provide some control of NO_3 build-up in Minnesota fish farms.

Future Management of Caland Pit Lake

The most significant result of this study was that the relatively small volume of the mixolimnion could have such a pronounced influence on the water quality of the very much greater volume comprising the monimolimnion. This has considerable implications for the management of these pit lakes.

Oxygen levels are a critical issue. It may be necessary to enhance dissolved oxygen levels in the mixolimnion through the implementation of aeration systems such as windmills, pumps, or fountains. A second strategy should be to limit input of organic material to the monimolimnion, possibly through the use of closed containment methodologies (Dodd 2000).

Continued filling of Caland pit lake could also have a devastating effect. If the input of fresh water cannot keep pace with increasing surface area, the mixolimnion volume will decrease, and the depth of the water column that is suitable for aquaculture will be severely reduced and may disappear entirely. This effect will accelerate once Caland pit lake joins with Hogarth pit lake. The sulfate level will also increase considerably after this union since Hogarth pit lake has a sulfate concentration of approximately 1,500 mg·L^{-1}, which is more than four times that of Caland pit lake (McNaughton 2001). The concentration of the outflow will therefore greatly exceed the Canadian water quality guidelines for drinking water for sulfate of 500 mg·L^{-1} (Health Canada 2008).

Other detrimental effects may occur downstream. Effluent from tailings areas at another mine site in northwestern Ontario has been documented as causing meromictic conditions downstream leading to oxygen depletion in the monimolimnion of receiving lakes containing brook trout (Haapa-aho 2004). Another concern is downstream effects on wild rice, *Zizania palustris* L., which is a commercial crop in this area (Aiken et al. 1988). In nearby Minnesota, Section 2024 of Chapter 7050, Waters of the State, states that industrial discharges of sulfate must not exceed 10.0 mg·L^{-1} because of the potential harm to wild rice stands (Minnesota Administrative Rules, 2010). Methods must be developed to reverse this situation. It may be feasible to reduce the sulfate concentration of drainage water and/or outflow

water. One possibility is to utilize wetlands to filter the runoff from the mine (Noller et al. 1994). Another possibility is to increase the volume of the mixolimnion by inputting freshwater from nearby lakes.

As an after note, detailed water quality monitoring and research on a variety of topics (toxicity, ground water sources, wetland effects) have continued on Caland pit lake since this initial study and are contained in M.Sc. theses (Vancook 2005; Goold 2008). These studies have shown that the prediction of a reduced mixolimnion is still valid, but the timing of this occurrence has changed. Originally a filling model suggested overflow into the Seine River would occur in 2030 (Vancook 2005), but this has been revised to between 2070 and 2100 (Jackson 2007). The loss of the mixolimnion will still occur, but not until a much later date. The fish farm also decreased production and eventually closed in 2006. The result has been that the water quality in Caland pit lake seems to be improving slowly. For example, both nitrate and potassium have declined in concentration in both the monimolimnion and mixolimnion (Fig. 5). Sulfate however remains a problem and is a current focus of research to reduce its concentration. In 2009, the Government of Ontario established the Steep Rock Rehabilitation Project which is to develop, over a three year period, a management plan that will rehabilitate the two pit lakes and ensure the protection of downstream waters. Environmental hazards and site management are to be properly addressed by 2032.

Conclusions

In this study meromictic conditions were demonstrated in a pit lake that was formerly used for mining iron. Freshwater inflow from surface waters was sufficient to develop a mixolimnion that in 1998 extended to a depth of 25 m. Below this was the monimolimnion that extended a further 155 m. The two layers were separated by a chemocline where cations and anions, dominated by sulfate, increased in concentration. A fish farm was established in the mixolimnion of the pit lake, and aquaculture effects in the upper layer impacted the water quality in the monimolimnion causing elevated levels of nutrients, including nitrogen, phosphorus, and potassium, and a lowering of oxygen to anaerobic concentrations. However, the effect of the mixolimnion on nutrient concentrations throughout the water column seems to be reversible. After the fish farm ceased operation, a gradual lowering of nutrients occurred in both the mixolimnion and the monimolimnion. Sulfate levels, on the other hand, have not shown the same trend of declining concentrations with time and remains a concern that needs to be remediated.

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