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The limnology of two proximal pit lakes after twenty years of intense flooding

by

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of the requirements for the degree of
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Abstract

Two adjacent pit-lakes, Caland and Hogarth, formed after the flooding of open pit iron mines near Atikokan, Ontario, Canada, were examined for their water quality relationships and potential to support aquaculture. Detailed sampling for physical and chemical parameters took place during 1998, 1999, and early 2000.

Since mining terminated in 1979, both pits have continued to fill with water from rainfall, snowmelt, and groundwater seepage, and, by 2000, had water depths in excess of 160 m. Water quality in both pit-lakes was largely dependent on proximal waste rock composition and surficial geology of the area. Limestone and dolostone deposits at the site countered production of acid from waste rock and resulted in neutral water pH's in Caland and Hogarth pit-lakes. Drainage was a major factor in distinguishing the water quality between the two pit-lakes. Inflow of freshwater was much greater into Caland and resulted in meromictic conditions with a well defined mixolimnion, chemocline and anoxic monimolimnion. Hogarth pit-lake had less inflow and the entire water column was sulfate-saline and aerobic. Statistically significant variations occurred between the two pit lakes for metal, anion, cation, conductivity, hardness, and dissolved oxygen concentrations. Variations in precipitation and evaporation between the two sampling years resulted in increased or decreased ion concentrations for the same seasons or depths within the two pit-lakes.

Caland pit-lake has supported an aquaculture operation for the production of rainbow trout (*Oncorhynchus mykiss*) since 1989. Production increased from 20,000 kg in 1998 to 163,325 kg in 1999. Nutrient loading from excess fish food and waste increased phosphorus and ammonia levels during 1999 and 2000. The expanding fish farm also decreased the volume of usable (>6.0 $mg \cdot l^{-1}$) dissolved oxygen. Hogarth pit-lake was devoid of aquatic life. Using water from Hogarth, standard LC_{50} tests showed 100% mortality of *Daphnia magna* at full strength concentrations. None of the tested metal parameters had levels that would be considered toxic and elevated salinity was suspected as the cause of mortality. Any future aquaculture in Hogarth pit-lake will require the use of species tolerant to such saline conditions.

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General Introduction

The closure of open-pit mines and their subsequent filling with both groundwater and surface water has produced many pit-lakes (Banks et al., 1997; Doyle and Runnells, 1997; Hamblin et al., 1997; Miller et al., 1996). Pit-lakes may act as discharge areas, recharge areas, or flow-through areas depending on where the pits were excavated within the regional flow system. The physical characteristics of pit-lakes differ from those of most natural lakes in that they are usually very deep and have low surface:depth ratios. One important factor influencing the water quality of pit-lakes is their physical limnology. Included are such issues as thermal and chemical stratification, turnover, time, and any development of anoxia with increasing depth (Wetzel, 1983). Other unique characteristics of pit-lakes include high surrounding walls that often generate a microclimate within the pit, thin sediment layers, substantial and rapid changes in water level, and high concentrations of dissolved material (Stevens and Lawrence, 1998).

The process of open pit mining produces newly scoured surfaces. These present an opportunity to study the effects of the chemically active rock walls, waste rock, and tailings on closed lake systems. Generally the chemical composition of the water depends on the minerals present in the exposed rock, their relative abundances, their reaction rates, the actual weathering reactions involved, and drainage conditions (Eugster and Hardie, 1978).

Pit-lake waters vary greatly in composition. They range in water quality from very polluted to sometimes serving as a drinking water supply in nearby communities. As concurred by Banks et al (1997), mine waters need not be perceived as problems, but can often be regarded as sources of drinking water or for industrial uses such as: sewage treatment; tanning; and industrial metals extraction. In addition some abandoned mine sites may serve as sites for inland aquaculture. This appeals to mining companies as a less costly alternative method for post operation remediation, but the utilization for aquaculture adds additional nutrients that may increase lake productivity in these unsettled environments.

In this study two proximal pit-lakes, Caland and Hogarth, located on the Canadian Shield near Atikokan, Ontario were examined. Caland and Hogarth pit-lakes formed in the east and middle arms respectively of the original Steep Rock Lake lake-bed. Caland pit was mined from 1960 to1977, and Hogarth pit from 1954 to 1977. After closure, the pits filled with water from rainfall, snowmelt and groundwater seepage reaching present depths greater than 160 meters. Limestone and dolostone deposits at the site counter production of acid from waste rock resulting in neutral water pH in both pit-lakes. Surficial geology is similar surrounding the two pit-lakes except for the sporadic outcrops of pyritic lenses at the Hogarth site. The drainage basin encompasses an area of 11.7 km^2 for Caland pit-lake and 5.1 km^2 for Hogarth pit-lake.

Caland pit-lake has an aquaculture operation (Snow Lake Fish Farm) which produces rainbow trout (*Oncorhynchus mykiss*). In 1998 fish density was 20,000 kg which increased due to a major expansion of the facility. By December 1999, fish numbers averaged 733,325 with a biomass 163,325 kg, and by August 2000 production reached 227,000 kg (Dave Lindsay (Snow Lake Fish Farm owner) pers comm, 2000). Chapman (1997) assessed the biological capacity of Caland pit-lake. It was determined that the increase in annual production to 227,000 kg of rainbow trout would be supported by the pit-lake. There were only six fish farms in Ontario which produced over 200,000 kg of fish in 1998 (Moccia and Bevan, 1999).

Curiously, Hogarth pit-lake, located approximately a kilometer from Caland pit-lake, is sulfate-saline. The potential for using this pit-lake which is high in dissolved solids for inland aquaculture is questionable.

The main objectives of the study were i) to determine the effects of surficial geology, climate and drainage on the water quality of the two proximal pit-lakes; and ii) to determine the influence of aquaculture on Caland pit-lake water quality, and the potential for aquaculture in Hogarth pit-lake.

1. The utilization of an abandoned open-pit mine on the Canadian Shield for inland aquaculture as a viable alternative for post-mining remediation

1.1 Introduction

Newly scoured surfaces present a unique opportunity to study the effects of chemically active rock walls, waste rock, and tailings on the water quality of flooded open pit mines. Generally the composition of the water depends on the minerals present in the exposed rock, their relative abundances, their reaction rates, the actual weathering reactions involved, and the drainage condition (Eugster and Hardie, 1978). Establishment of aquaculture in these abandoned open pit mines adds additional chemical nutrients that may increase lake productivity in these unsettled environments.

In northern Minnesota's Mesabi Range, aquaculture operations were established in several abandoned open pits that had moderately hard water (Axler et al., 1996). In Quebec, a rainbow trout rearing facility was located at Lac Du Passage and was also characterized by alkaline water (Cornel and Whoriskey, 1993). The acid buffering capacity of these flooded pits can therefore be suitable for fish cultivation. Unfortunately a problem that often arises from the actual aquaculture operation is the development of anoxic conditions in the hypolimnion as chemical and biological oxidation and microbial breakdown of fish food and feces consume oxygen (Axler et al., 1992). In the absence of oxygen, phosphorus (P) is released at the sediment - water interface and mixed throughout the water column at spring and fall turnover, producing eutrophic conditions (Mortimer, 1941). Any discharge from the pits will then adversely affect the water quality of the receiving water body. This was essentially the situation in Minnesota and the inability of the aquaculture company to meet discharge standards ultimately caused its bankruptcy (Axler et al., 1992). Non compliance to environmental water quality regulations has therefore limited the use of open pits for aquaculture. Nevertheless, the premise of utilizing an abandoned mine site for aquaculture remains attractive from both an economic and social sense. The challenge is to devise a set of criteria for identifying suitable open pits for this purpose and to develop the needed techniques that will limit any detrimental effects on water quality.

A fish farm, located in a flooded open pit on the Canadian Shield, provided the opportunity to examine the implications of fish farming as a remediation alternative. The objectives of this study were to determine i) if a substantial increase in production would influence water quality in the pit-lake over a two year period; ii) if acid rock drainage (ARD) persisted from the exposed waste rock and if it and/or micro-climate affected the water quality; and iii) if the fish farm influenced the historical water chemistry of the flooded pit.

1.2 Method

1.2.1 Study Site

The study site was located at the southern margin of the granite-greenstone Wabigoon Subprovince of the Superior Province of the Canadian Shield (Ontario Ministry of Northern Development and Mines, 1994). The Steep Rock area, near Atikokan, Ontario (Figure 1.1) contains Archean metavolcanic, metasedimentary and intrusive rocks which were displaced by a series of faults (Shklanka, 1972). The occurrence of iron bearing rocks in the Atikokan region was first recognized by Smyth in 1891, but it was not until 1930 that the ore bodies were located below Steep Rock Lake (Steep Rock Mines, 1943). Since the iron ore body was located under Steep Rock Lake, the lake had to be drained. This was accomplished by diverting the inlet, the Seine River, to flow through Finlayson Lake and installing a series of dams. This diversion was completed in 1943, and the net effect moved the Seine River two miles westward for eight miles of its course. The original depth of Steep Rock Lake was approximately 70m (Steep Rock Resources Inc., 1986). The Middle and East Arms of the lake were pumped dry and about 205 million m^3 of overburden were removed. The Caland deposit was located in the East Arm of Steep Rock Lake (48° 48'N, 91°39'W). In 1960 ore extraction commenced in the Caland zone. Total ore mined between 1960 and 1977, when it closed, was 33 million long tons. After mining ceased, the rights to the land reverted to the Government of Ontario and the pit began to fill with water from rainfall, runoff, and groundwater seepage.

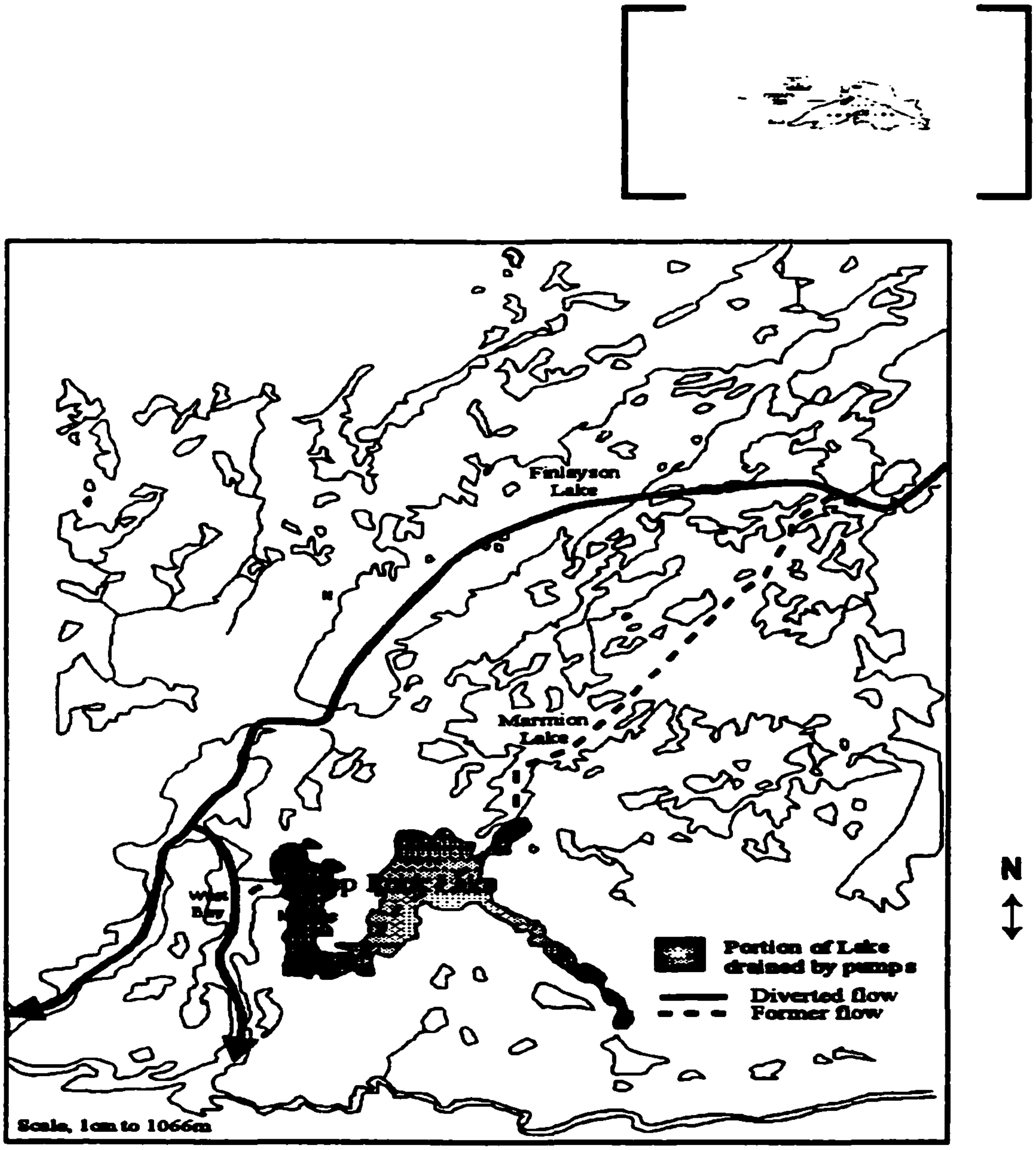

Figure 1.1: Location map of Steep Rock Lake near Atikokan, Ontario. Caland pit-lake is located in the former East Arm of Steep Rock Lake, and Hogarth pit-lake is located in the northern section of the Middle Arm. (Modified from Pye (1968)).

The drainage basin for Caland pit-lake encompasses an area of 11.7 km^2 (Steep Rock Iron Mines Ltd. 1978). As of 1999, Caland pit-lake depth and area were approximately 180 m and 120 ha, respectively. Caland pit-lake has a bowl/cone shaped basin morphology with a very low surface area to depth ratio. This pit-lake is steep sided and well sheltered from winds thus promoting its meromictic environment.

Snow Lake Fish Farm commenced its operations in 1988, but actual farming began with chinook salmon (*Oncorhynchus tshawytscha*) in 1989. 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, due to damage from ice and storms, rainbow trout production decreased to 36,000 kg. In 1997 one-half of the remaining rainbow trout were sold, leaving roughly 20,000 kg most of which were unmarketable due to their undesirable size. 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 (D. Lindsay, pers comm, 2000).

Snow Lake Fish Farm uses cage culture for its rainbow trout operation. The normal procedure is to place rainbow trout fingerlings in 7.6 m diameter x 10 m depth cages in the spring where they remain until 10 cm in length. The fish are then transferred first to cages 15 m diameter x 15 m depth, and finally to 23 m diameter x 20 m depth cages where they are grown to market size (1-2 kg). The larger cages have a capacity of 20,000 kg in the summer and 40,000 kg in winter. The fish are raised throughout the year but growth occurs primarily in the summer. During 1998, the total quantity of fish food utilized was approximately 146,000 kg versus 214,000 kg used by October 1999. Processing of the fish was done on site in a floating building until 1999 when a processing plant was built in Atikokan.

The pit-lake mixolimnion contained a healthy population of *Daphnia* sp., several wild northern pike (*Esox lucius*), and schools of spottail shiners (*Notropis hudsonius*).

1.2.2 Field Procedures

For the duration of 1998 and 1999 three sampling stations were selected in Caland Pit. One station (180 m depth) was situated at the fish cages, a second station (120 m depth) was established northwest of the cages partially sheltered behind the island, and the third (90 m depth) was closer to the east shore (Figure 1.2). Water was sampled at each of the stations in the epilimnion (1 m), thermocline, hypolimnion (30 m) and 1 m off-bottom. Water was collected and sampled six times in 1998: monthly from June to September; before ice cover (early November); and through the ice (late February, 1999). The second year of sampling commenced after ice-out in mid May, then monthly from June to September, early November, and through the ice in March of 2000, totalling seven trips.

Figure 1.2: Site location map. Sampling occurred at three water chemistry stations in Caland pit-lake, Ontario. Station A was beside the fish cages, 180m deep, B was 90m deep, and C was 120m deep. Four drainages (north, east, south, and west) were also sampled.

Lake water was collected at each station using a Kemmerer Bottle lowered with calibrated rope. Secchi depth was measured using a secchi disk with calibrated line. Time of day, air temperature, and cloud cover were recorded, and wind speed estimated. Dissolved oxygen (D.O.) and temperature profiles were obtained from each station to the cable depth of 50 meters using Yellow Springs Instrument (YSI) Model #51B. Estimates of off-bottom D.O. and temperature were obtained by placing the YSI probe into the contained water sample in the Kemmerer bottle, thereby limiting exposure to atmospheric influences. Drainage water was also collected during the ice-free periods (totalling 11 sampling trips). The drainages were located at the north, east, south, and west of the pit. Drainage water was collected by surface grab method using a polyethelene bottle.

Each water sample was divided into three separate volumes before being transported back to the Lakehead University Environmental Lab (LUEL) for analysis. The three separate volumes were: 250 ml preserved with 1.0 ml of HNO_3 (conc) for total metal determinations; 250 ml preserved with 1.0 ml H_2SO_4 for total kjeldahl nitrogen, total phosphorus, and free ammonium; and 1 L unpreserved for total suspended and total dissolved solids, alkalinity, conductivity, dissolved organic and inorganic carbons, silicates, cations, and anions. Field sampling quality controls included 1 field blank, 1 travel blank, and 1 duplicate sample taken each sampling trip. Water from each sample was cooled on ice prior to shipment back to LUEL. Table 1.1 lists the physical and chemical factors examined at each sampling station.

Table-1.1: Chemical and physical factors examined at each sampling station in Caland and Hogarth pit-lakes, Ontario.

Chemical factors		
Water chemistry:		
	alkalinity	pH
	conductivity	hardness
	chloride	nitrate
	sulfate	total kjeldahl nitrogen
	total phosphorus	aluminium
	boron	beryllium
	cadmium	cobalt
	chromium	molybdenum
	nickel	lead
	vanadium	free ammonium
	reactive silicates	calcium
	copper	iron
	potassium	magnesium
	manganese	sodium
	strontium	zinc
	dissolved oxygen	total suspended solids
	total dissolved solids	dissolved organic carbon
	dissolved inorganic carbon	
Physical factors:		
	water depth	water temperature
	secchi	cloud cover
	sampling time	air temperature

1.2.3 Analytical Procedures

All in-lab analysis 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, then a repeat of the succeeding field sample. The QA/QC and the Repeat steps were repeated after every 10 field samples. Each of the tests followed LUEL Standard Operating Procedures (LUEL, 1998, 1999, 2000) which were modified from Standard Methods for the Examination of Water and Wastewater 18th ed. by Greenberg et al, 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 $uS{\cdot}cm^{-1}$.

Total suspended (TSS) and total dissolved (TDS) solids were determined gravimetrically and reported as weight/volume of sample. TSS refers to all material (residue, particulate) which was removed by filtration through a 0.45 um glass fibre filter. The material on the filter was dried at 103°C. The TDS fraction measured the amount of dissolved material that passed through a 0.45 um filter. This material collected in beakers was dried at 103°C.The filters and beakers were desiccated for 12 hours, then final weights obtained to 5 decimal places.

An aliquot of the unused portion of the filtered TDS sample was used to determine the anions SO_4, Cl, and NO_3 concentrations using a Dionex DX-120 Ion Chromatograph (IC) in conjunction with an AS40 automated sampler.

Dissolved inorganic (DIC) and dissolved organic (DOC) carbon 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 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 a 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 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 (Greenberg et. al., 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 660nm. The absorbance was proportional to the reactive silicate concentration in the original sample.

Total Phosphorus and Total Kjeldahl Nitrogen (TKN) were digested then analysed using a colorimetric determination on the Skalar Autoanalyser system. Before analysis, samples were treated with sulphuric 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 (Greenberg et. al., 1992) reaction. Total Phosphorus was simultaneously carried out on the same sample aliquot. The digestion converted all forms of phosphorus to orthophosphate which was determined colorimetrically at 880nm based on the ascorbic acid procedure (Greenberg et. al., 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 monochloriamine 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 -24 hours resulting in a two fold concentration of the sample according to MOEE (1994). The samples were analysed using the Jarrell Ash Inductively Coupled Argon Plasma 9000 Spectrometer (ICP). The dissolved cations (Ca, Mg, Na, and K) were analysed within four days of sampling, also on the ICP.

Total hardness, expressed as calcium carbonate in $mg \cdot l^{-1}$, is defined as the sum of the calcium and magnesium concentrations. The preferred method to compute hardness is taken from the results of separate ion determinations of calcium and magnesium in the calculation: Hardness ($mg \cdot l^{-1}$ CaCO3)= 2.497 x (Ca($mg \cdot l^{-1}$)) + 4.118 x (Mg($mg \cdot l^{-1}$)) (Greenberg et al. 1992).

Sulfide was analyzed based on a method outlined by EPA (1988) using the Cary SE UV-Vis-NIR Varian Spectrophotometer.

1.2.4 Data Analysis

A multivariate computer program from the Statistical Package for Social Sciences (SPSS) was used for the data analysis. The water chemistry (pit-water and drainage) was summarized using means and standard deviations. The pit-lake water chemistry was a combined mean of the three sampling stations. To test for seasonal trends, a Repeated Measures Anova was used for each of the 24 water chemistry parameters (parameters that were below method detection limits (<MDL) for greater than 50% of the measures were removed from the data set). Data collected from each station was entered into the program.The Repeated Measures Anova also allowed testing for significant ($p<0.05$) differences between the two sampling years, as well as depths. Due to this program requiring a balanced input, data collected in May 1999 was eliminated from the set. A hierarchal cluster analysis was used on the same data set to group the variables in 1998 and 1999. The division point for the clusters was determined at a Dissimilarity of 18.5.

Repeated measures was also used to determine water chemistry changes by comparing data collected by the Ministry of the Environment in June 1992, to the data sets for June from this study. The comparable depths used were the epilimnion (1 m), thermocline, and hypolimnion (30 m). The off-bottom data were not available in the 1992 data set.

1.3 Results

1.3.1 Water Chemistry

The mean and standard deviation for 24 water chemistry variables, separated by depth, with six replicates for 1998 and seven replicates for 1999 are shown in Table 1.2 and 1.3, respectively. The pit-lake water remained above neutral throughout the water column and was well buffered with average alkalinity greater than 120 $mgCaCO_3 \cdot l^{-1}$. The conductivity reflected a meromictic lake. The chemocline was at approximately 23 m, yet appeared to rise in parallel with any rise in surface level of the pit-lake. Most of the water chemistry parameters were typical of a meromictic lake with elevated concentrations below the chemocline for: alkalinity, DIC, TDS, NO_3, SO_4, Ca, Mg, hardness, Na, Cl, K, Fe, Mn, Sr, and SiO_2 (Table 1.2, 1.3). Hydrogen sulfide was less than 0.1 $mg \cdot l^{-1}$. The dissolved oxygen and temperature profiles were also indicative of a meromictic environment with oxygen depletion and eventual temperature increases below the chemocline (Figure 1.3). The oxygen depletions may have been occurring above the actual chemocline, that is just below the deepest fish pens.

In the mixolimnion, the dissolved oxygen profile in August 1998 had a slightly negative heterograde shape which was more distinct in August 1999 (Figure 1.3). With turnover, the oxygen concentrations replenished in the winter to a depth of 20 m (Figure 1.3). During the winter, ice above the fish pens did not form due to the constant fish movement but was greater than 0.5 m thick elsewhere on the lake. The 1998 and 1999 August temperature

Table 1.2: The 1998 water chemistry means and standard deviations for Caland pit-lake. Concentrations are in mg·l^{-1} except for alkalinity (mgCaCO$_3$·l^{-1}), conductivity (uS·cm^{-1}), hardness (calc.), pH and temperature (^{0}C). The ^{abc} indicates significant ($p<0.05$) differences between the two years of study (^{a}), with season (^{b}), or with depth (^{c}).

	Epilimnion		Thermocline		Hypolimnion		Off-bottom	
	mean	std.dev.	mean	std.dev.	mean	std.dev.	mean	std.dev.
Alkalinity ^{abc}	125.72	5.22	129.51	4.60	167.95	5.29	172.20	3.67
Conductivity ^{abc}	702.61	33.47	718.83	30.85	1056.28	87.87	1131.72	78.68
DIC ^{abc}	32.33	4.36	33.70	7.41	43.73	5.99	46.13	4.28
DOC ^{abc}	3.74	1.03	3.74	1.13	3.44	1.15	3.49	1.06
Hardness ^{abc}	360.36	10.59	370.74	16.84	593.19	24.57	644.18	21.44
Cl^- ^{abc}	7.53	0.76	7.69	0.81	12.95	1.64	13.54	1.84
NO_3^- ^{abc}	2.44	0.45	2.58	0.45	4.40	1.24	4.20	1.26
SO_4^- ^{abc}	243.19	23.37	242.84	22.79	442.50	58.50	497.23	67.28
Ca ^{abc}	83.93	3.17	86.93	4.70	130.61	5.20	139.80	4.92
K ^{ac}	4.33	0.41	4.34	0.40	5.27	0.36	5.83	1.52
Mg ^{abc}	36.62	0.74	37.31	1.30	64.84	2.97	71.64	2.31
Na ^{ac}	11.96	0.36	12.22	0.58	18.15	0.68	19.02	0.57
B ^{ac}	0.06	0.019	0.066	0.03	0.073	0.033	0.073	0.032
Fe	0.019	0.01	0.016	0.007	0.025	0.039	0.035	0.032
Mn ^{abc}	0.006	0.004	0.005	0.002	0.065	0.024	0.184	0.066
Sr ^{bc}	0.621	0.023	0.633	0.026	0.966	0.061	1.02	0.12
NH_4 ^{abc}	0.049	0.053	0.059	0.054	0.027	0.023	0.03	0.02
pH ^{bc}	7.90	0.35	7.58	0.35	7.00	0.14	6.98	0.12
SiO_2 ^{abc}	2.57	0.91	3.46	1.33	9.78	2.51	10.79	2.18
TKN ^{c}	0.233	0.149	0.257	0.1	0.293	0.107	0.249	0.123
Total - P ^{abc}	0.009	0.014	0.008	0.006	0.006	0.003	0.006	0.003
TDS ^{ac}	479.16	22.70	491.79	25.39	770.09	39.95	859.83	72.24
Temp (^{0}C) ^{bc}	15.77	8.29	13.36	5.32	5.33	0.27	6.03	0.12
D.O. ^{abc}	9.26	2.50	9.54	2.28	0.55	0.19	0.30	0.13

Table 1.3: The 1999 water chemistry means and standard deviations for Caland pit-lake. Concentrations are in mg·l^{-1}except for alkalinity (mgCaCO$_3$·l^{-1}), conductivity (uS·cm^{-1}), hardness (calc.), pH and temperature (^{0}C). The ^{abc} indicates significant ($p<0.05$) differences between the two years of study (^{a}), with season (^{b}), or with depth (^{c}).

	Epilimnion		Thermocline		Hypolimnion		Off-bottom	
	mean	std.dev.	mean	std.dev.	mean	std.dev.	mean	std.dev.
Alkalinity ^{abc}	124.80	11.82	125.61	9.38	164.41	8.78	173.57	4.48
Conductivity ^{abc}	604.50	25.74	620.50	21.41	890.90	63.25	1008.10	25.68
DIC ^{abc}	27.08	2.90	28.73	2.25	39.66	3.22	42.35	3.96
DOC ^{abc}	3.58	1.49	3.46	1.48	3.23	1.56	3.09	1.40
Hardness ^{abc}	334.76	14.43	343.69	12.72	544.59	43.91	629.76	25.09
Cl$^-$ ^{abc}	6.30	0.37	6.39	0.29	10.44	0.99	11.50	0.69
NO$_3^-$ ^{abc}	1.92	0.11	1.98	0.09	3.21	0.36	2.83	0.34
SO$_4^-$ ^{abc}	217.18	28.88	222.61	26.74	348.44	34.57	417.28	28.04
Ca ^{abc}	78.18	3.16	80.59	2.89	120.88	8.67	136.07	6.08
K ^{ac}	3.94	0.37	3.97	0.42	4.88	0.38	5.19	0.37
Mg ^{abc}	33.89	1.68	34.60	1.45	58.95	5.48	70.42	2.55
Na ^{ac}	11.14	0.34	11.40	0.30	16.78	1.14	18.67	0.64
B ^{ac}	0.128	0.084	0.162	0.114	0.161	0.091	0.15	0.077
Fe	0.02	0.009	0.017	0.008	0.018	0.006	0.045	0.038
Mn ^{abc}	0.007	0.001	0.006	0.002	0.067	0.022	0.326	0.08
Sr ^{bc}	0.599	0.038	0.616	0.031	0.92	0.066	1.05	0.052
NH$_4$ ^{abc}	0.135	0.08	0.17	0.07	0.061	0.061	0.073	0.054
pH ^{bc}	7.76	0.46	7.49	0.27	7.13	0.21	7.09	0.18
SIO$_2$ ^{abc}	2.91	0.61	3.11	0.51	8.31	1.24	9.07	1.36
TKN ^{c}	0.384	0.084	0.452	0.11	0.337	0.093	0.316	0.079
Total - P ^{abc}	0.011	0.005	0.014	0.009	0.01	0.005	0.02	0.012
TDS ^{ac}	450.57	23.67	459.27	20.42	707.74	59.46	838.39	34.39
Temp (^{0}C) ^{bc}	14.00	7.48	11.16	5.91	5.65	0.22	6.05	0.22
D.O. ^{abc}	9.69	0.76	8.18	2.19	0.38	0.16	0.27	0.02

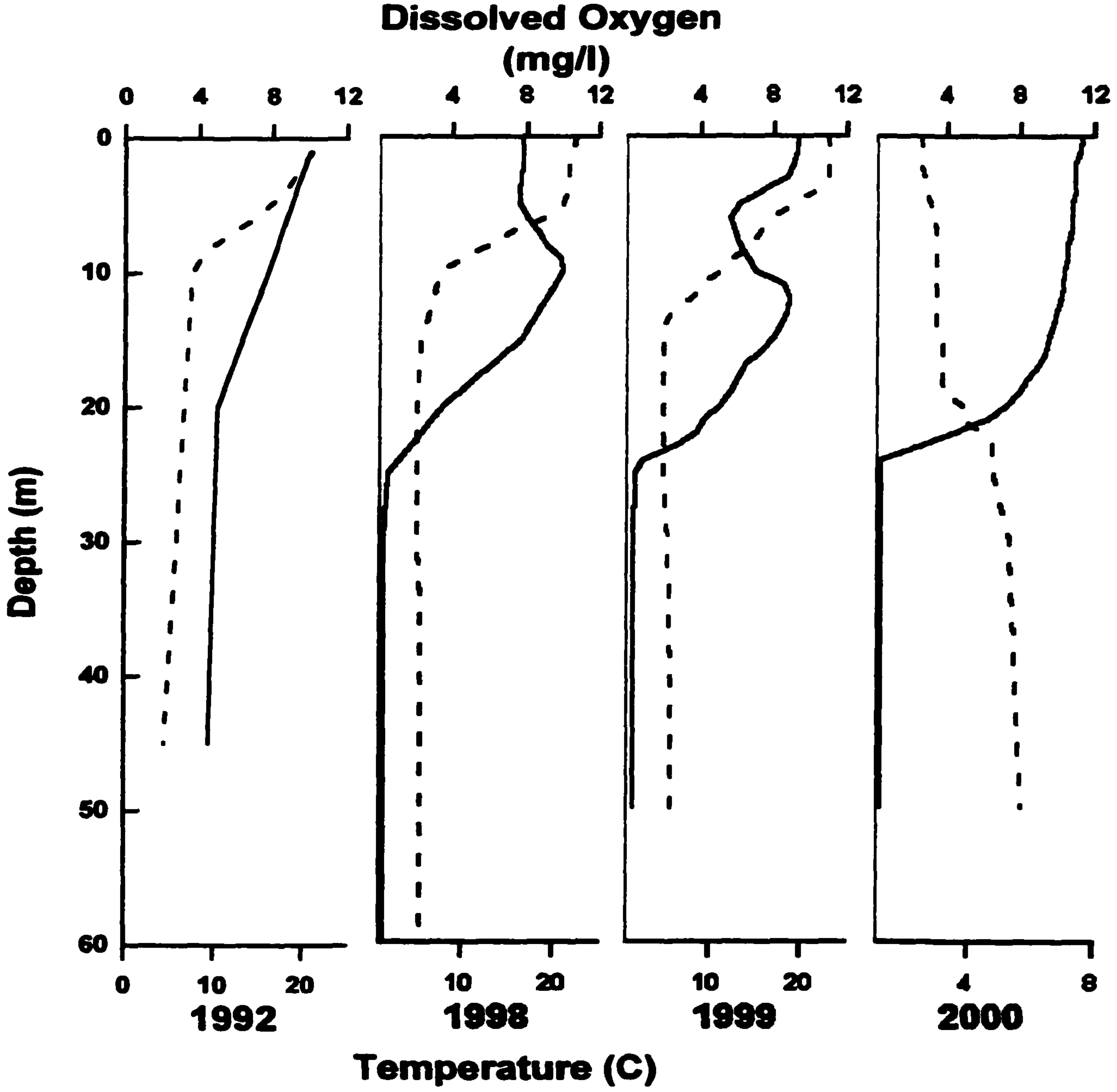

Figure 1.3: Dissolved oxygen (—) and temperature (---) profiles for June 1992 (OMOE), August 1998 and 1999, and winter 2000. With the exception of the 1992 profiles, the profiles are averages of the three sampling stations in Caland pit-lake, Ontario.

profiles also showed thermal stratification in the mixolimnion with a defined thermocline and a hypolimnion which commenced at 15 m (Figure 1.3).

1.3.2 Variation between years, seasons and depth

The Repeated Measures Anova identified variations between 1998 and 1999, among seasons, and among depths (Table 1.2, 1.3). Most parameters significantly varied between the two years of study: alkalinity, conductivity, DIC, Cl, NO_3, Ca, Mn, SiO_2, DOC, SO_4, hardness, Mg, NH_4, D.O., K, Na, B, TDS, and Total-P. Several of these parameters decreased in concentration in 1999 (Table 1.2, 1.3, Figure 1. 4, Appendix B). Alkalinity, D.O., and K decreased in the epilimnion and thermocline. Hardness, Mg, and Na similarly decreased in the two prior mentioned depths as well as in the hypolimnion. SO_4 decreased only off-bottom. Also, several parameters decreased at each measured depth: conductivity, Cl, NO_3, Ca, DIC, DOC, and TDS. Conversely, several parameters increased in concentration during the second year of study. Mn increased in the off-bottom depth; SiO_2 and NH_4 increased in shallower depths; and, B and Total-P increased at each depth (Table 1.2, 1.3, Figure 1.4, Appendix B).

Variations with depth were significant for every parameter except Fe (Table 1.2, 1.3, Figure 1.4). A hierarchal cluster analysis was used to illustrate the variations with depth based on Pearson correlation. The resultant 1998 and 1999 hierarchal cluster analysis were very similar (Figure 1.5). In 1998, three clusters were clearly identified, which were

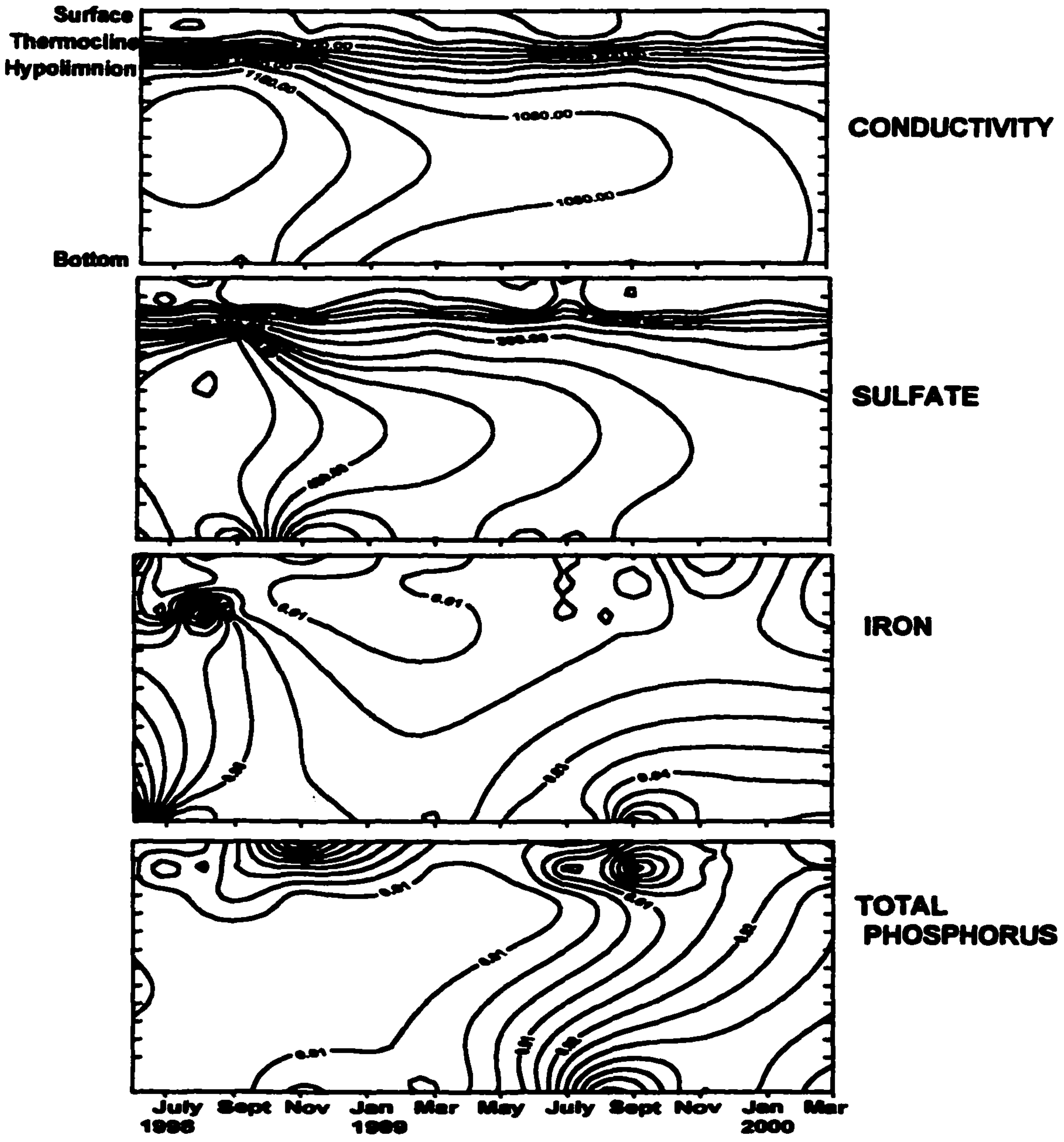

Figure 1.4: Conductivity, sulfate, iron, and total phosphorus average concentrations of the three sites at represented depths shown for each sampling occasion. Each of the parameters are in mg·l^{-1} except conductivity (uS·cm^{-1}).

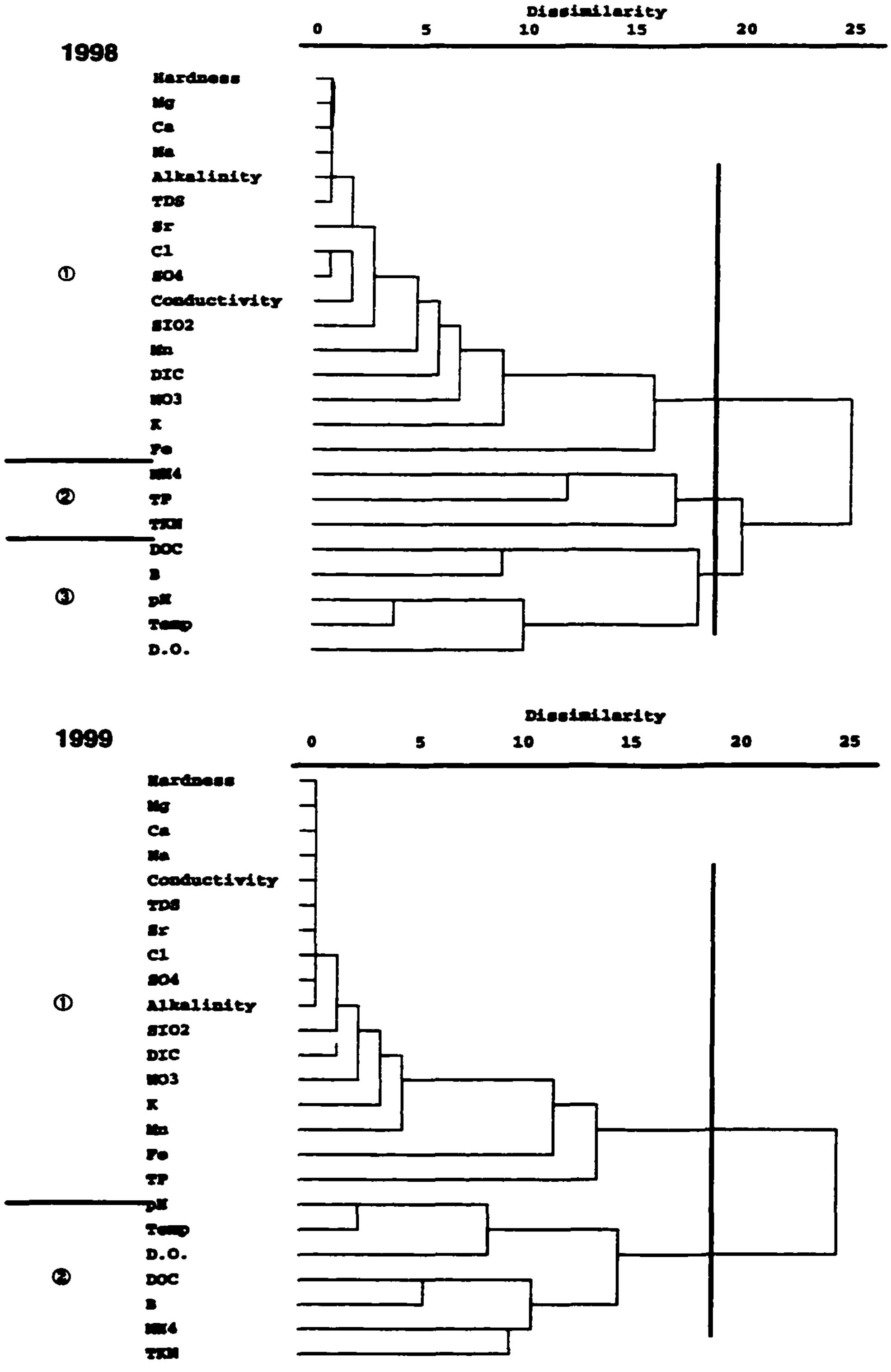

Figure 1.5: Hierarchal cluster analysis using the Pearson correlation average linkage between groups for the water chemistry parameters measured in 1998 and 1999 in Caland pit-lake, Ontario.

further simplified to two clusters in 1999 (Figure 1.5). The first cluster for 1998 and 1999 represented water chemistry parameters typical of a meromictic lake with elevated concentrations below the chemocline for: hardness, conductivity, alkalinity, Mg, Ca, Na, TDS, Sr, Cl, SO_4, SiO_2, Fe, Mn, DIC, NO_3, and K (Figure 1.4, 1.5, Appendix B). Although Fe was not significant for depth, its monimolimnion concentration was higher than the mixolimnion (Figure 1.4). In 1998, the second cluster consisted of NH_4, TKN, and Total-P. NH_4 and Total-P levels were generally <MDL (0.015 $mg \cdot l^{-1}$ and 0.005 $mg \cdot l^{-1}$, respectively). In 1999, NH_4, TKN, and Total-P increased in the shallower waters, especially the thermocline. Since Total-P concentrations also increased off-bottom in 1999 it was included in cluster 1 with parameters typical of meromictic lakes (Figure 1.5). The third cluster for 1998 consists of parameters with either no discernable variation with depth such as B and DOC, or those with higher concentrations in the mixolimnion including pH, temperature, and D.O. (Figure 1.5). (TKN and NH_4 were included in this grouping in 1999 (Figure 1.5)). Temperature, pH, and D.O. changed significantly with depth as well as with season in the shallower waters (Table 1.2,1.3, Figure 1.3, Appendix B).

With the exception of Fe, TKN, Na, K, TDS, and B, all other parameters significantly varied with season (Table 1.2, 1.3). The spatial and temporal distribution of Na and K in lakes was uniform and demonstrated very little seasonal variation, indicative of the conservative nature of these ions (Stangenberg-Oporowska, 1967). The most common influence in the epilimnion and thermocline which affected each of the parameters was the onset of winter: water temperature and pH decreased; and each of the other parameters increased in concentration (Table 1.2, 1.3, Figure 1.4, Appendix B). DOC also showed several peaks throughout the sampling season which coincided with rainfalls.

1.3.3 Effect from acid rock drainage or local climate

Two overland drainages (north and east) and two ponds (south and west) were sampled during the ice-free periods. Means and standard deviation for 21 of the water chemistry parameters measured for the four drainages are shown in Tables 1.4 and 1.5. Representative parameters comparing between the 1998 and 1999 averaged concentrations are presented in Figure 1.6. Sampled drainages were neutral (Table 1.4, 1.5, Figure 1.6) revealing no ARD. The drainage water chemistry was relatively similar between the two years of study despite the difference in climate. In 1998, with overall less precipitation (Table 1.6) the surface water levels of the two sampled ponds were lowered and the overland drainages were moderately slowed. The high standard deviations in the drainage water chemistry reflect the variations in localized climate at time of sampling. Inflow as well as direct precipitation increased the pit-lake surface water level by approximately 5 m over the 2-year study period.

1.3.4 Water chemistry changes since 1992

Repeated Measures Anova identified variations between June1992, 1998, and 1999, and with depth (Figure 1.7, Appendix C). In June 1992, the chemocline was not as distinct as

Table 1.4: The 1998 drainage water chemistry means and standard deviations for Caland pit-lake. Concentrations are in mg·l^{-1} except for alkalinity (mgCaCO$_3$·l^{-1}), conductivity (uS·cm^{-1}), hardness (calc.), and pH.

	North		South		West		East	
	mean	std.dev.	mean	std.dev.	mean	std.dev.	mean	std.dev.
Alkalinity	119.38	6.97	122.22	6.81	36.40	1.48	117.60	10.15
Conductivity	263.60	9.71	1639.20	86.11	1728.80	2359.88	278.00	27.81
DIC	27.66	8.05	34.28	3.38	9.77	1.70	29.54	2.59
DOC	4.64	1.18	5.98	0.89	3.24	0.87	4.25	0.85
Hardness	133.10	8.61	1049.09	56.86	350.51	17.43	130.83	16.74
Cl^-	3.34	0.74	3.29	1.07	3.19	4.61	7.66	1.23
NO_3^-	0.03	0.04	0.01	0.00	0.01	0.00	0.11	0.06
SO_4^-	11.74	6.45	867.25	468.45	851.16	1215.89	28.10	34.03
Ca	41.07	2.85	163.47	4.49	93.69	5.19	37.56	5.34
K	2.54	0.58	4.64	0.76	2.48	0.63	1.60	0.21
Mg	7.42	0.38	155.57	11.43	28.31	1.25	9.00	0.86
Na	4.28	0.10	12.43	0.79	2.10	0.09	7.95	0.23
Al	0.17	0.12	0.09	0.03			0.06	0.01
B	0.07	.	0.22	0.15	0.08	0.02	0.10	0.03
Fe	0.25	0.15	0.06	0.01	0.07	0.03	0.07	0.05
Mn	0.02	0.01	0.13	0.10	0.24	0.23	0.01	0.01
Sr	0.111	0.005	0.63	0.027	0.231	0.008	0.139	0.012
pH	8.11	0.21	8.05	0.21	6.97	0.48	7.51	0.11
SiO_2	12.04	1.87	1.24	0.89	6.04	0.96	5.98	0.80
TKN	0.14	0.04	0.24	0.07	0.25	0.19	0.12	0.06
TDS	154.92	17.89	1420.48	111.56	508.56	59.84	192.88	68.92

Table 1.5: The 1999 drainage water chemistry means and standard deviations for Caland pit-lake. Concentrations are in mg·l^{-1} except alkalinity (mgCaCO$_3$·l^{-1}), conductivity (uS·cm^{-1}), hardness (calc.), and pH.

	North		South		West		East	
	mean	std.dev.	mean	std.dev.	mean	std.dev.	mean	std.dev.
Alkalinity	124.73	12.06	139.37	13.49	40.32	4.59	115.37	10.64
Conductivity	241.67	21.00	1134.33	115.34	499.33	90.68	240.50	20.85
DIC	27.69	3.09	25.29	12.69	9.13	1.25	25.99	3.48
DOC	4.49	1.57	7.15	1.48	3.59	0.98	4.59	1.19
Hardness	134.18	12.26	742.90	54.37	278.24	47.31	122.19	6.89
Cl^-	5.58	7.85	2.08	0.78	0.63	0.16	6.48	0.29
NO_3^-	0.05	0.07	0.01	0.00	0.01	0.01	0.08	0.02
SO_4^-	43.72	78.89	573.28	46.83	237.60	52.43	19.10	27.49
Ca	41.79	4.26	130.30	6.21	75.98	10.87	35.28	2.72
K	1.77	0.48	3.95	0.52	2.00	0.58	1.95	0.34
Mg	7.24	0.42	101.39	10.21	21.50	5.05	8.28	0.18
Na	4.01	0.28	8.83	0.72	1.88	0.20	7.39	0.26
Al	0.26	0.12	0.19	.	0.07	0.02	0.06	0.00
B	0.15	0.07	0.12	0.04	0.15	0.02	0.15	0.06
Fe	0.26	0.12	0.09	0.06	0.06	0.03	0.08	0.03
Mn	0.03	0.01	0.31	0.20	0.91	0.49	0.01	0.00
Sr	0.111	0.012	0.43	0.054	0.2	0.033	0.138	0.013
pH	8.19	0.15	8.12	0.17	6.84	0.24	7.83	0.27
SiO_2	7.42	1.15	3.82	1.00	4.91	0.90	3.58	0.79
TKN	0.12	0.06	0.33	0.08	0.18	0.05	0.16	0.09
TDS	153.07	13.08	1002.08	123.61	554.36	363.75	145.76	18.24

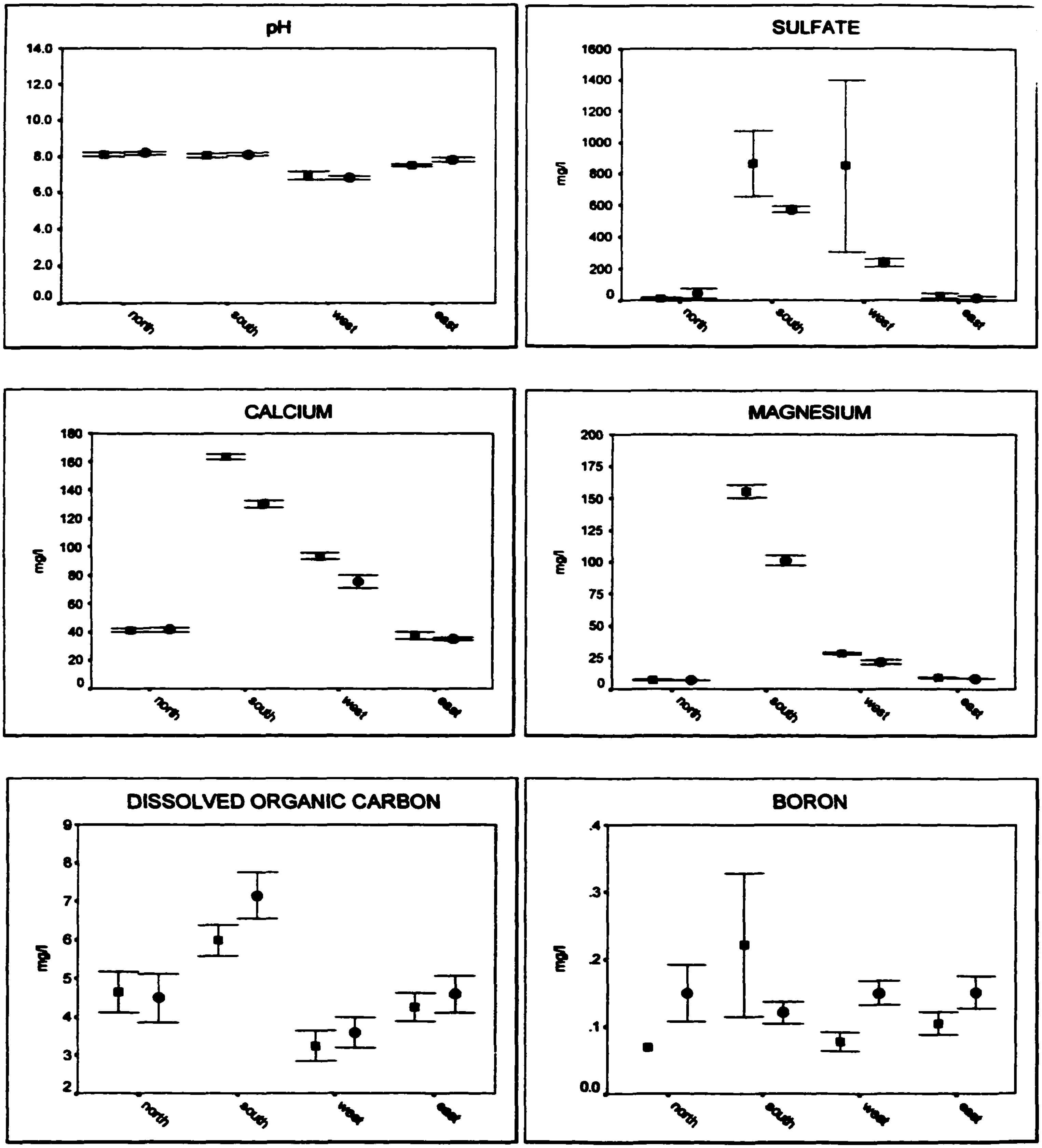

Figure 1.6. The mean and standard error for representative 1998 (■) and 1999 (●) drainage water chemistry parameters entering Caland pit-lake, Ontario.

Table 1.6: Mean monthly temperatures and total precipitation for 1997, 1998, and 1999. This data was supplied from Environment Canada's Atikokan-Mine Centre weather station near both Caland and Hogarth pit-lakes, Ontario.

	Mean Temperature (°C)			Total Precipitation (mm)		
	1997	1998	1999	1997	1998	1999
January	-16.3	-10.9	-15.9	29.6	22.6	11.6
February	-11.2	-2.4	-7.2	14.4	37.2	25.4
March	-6.6	-3.1	-3.1	36.6	20.4	60.2
April	2.3	6.5	5.6	41.8	10.8	34.2
May	8.1	13.2	12.3	38.6	85.6	127.5
June	17.6	15.6	16.7	64.6	73.8	102.4
July	18.6	18.4	19.9	66.6	67.4	209.0
August	16.9	19.6	17.0	29.4	43.2	66.6
September	14.1	14.5	11.6	126.8	51.8	174.6
October	5.7	7.1	4.8	56.2	113.6	38.2
November	-4.5	-2.0	0.2	33.0	48.2	5.2
December	-6.6	-10.6	-9.2	13.0	23.8	34.2
Total	3.2	5.5	4.4	550.6	598.4	889.1

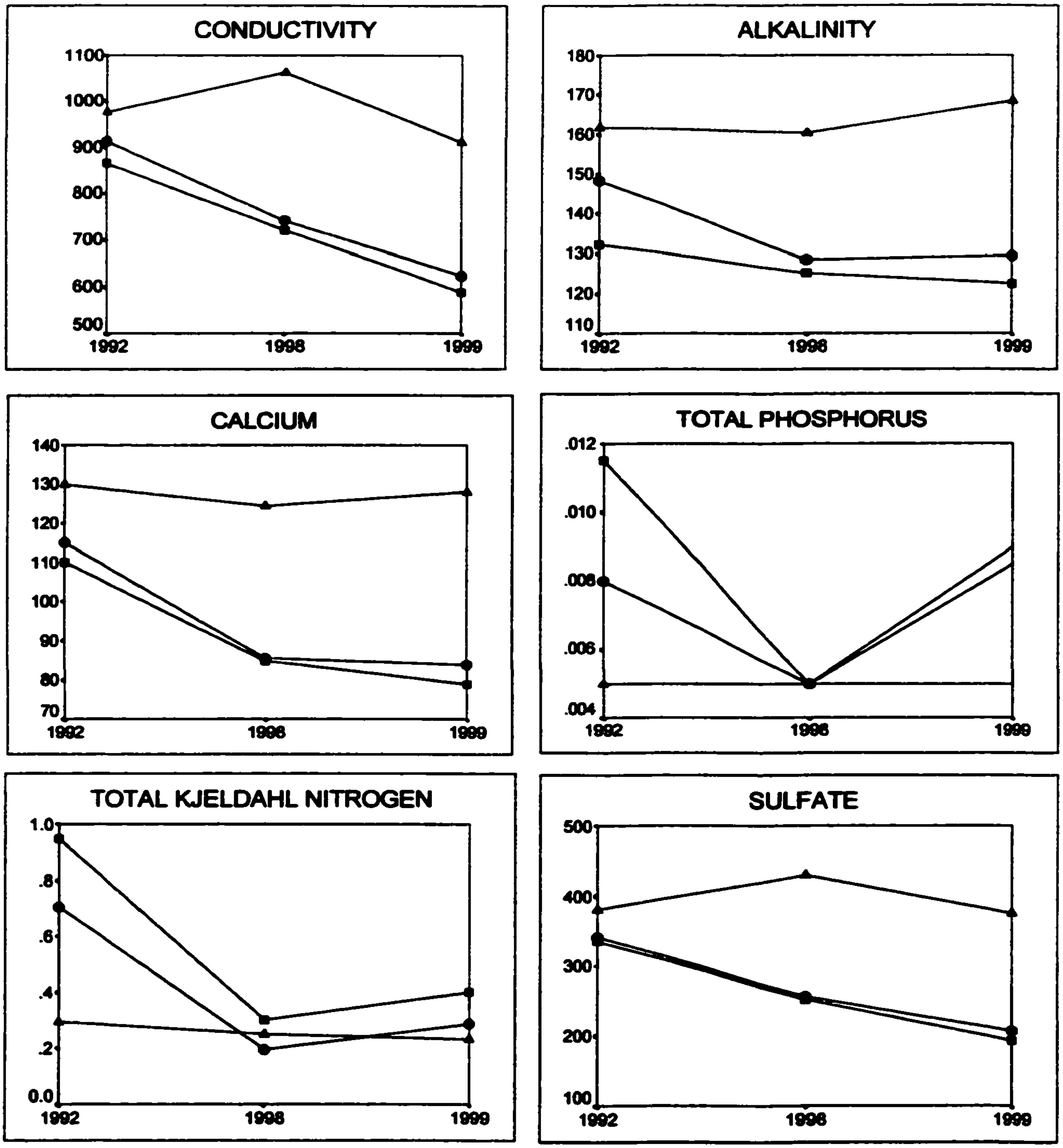

Figure 1.7: A water chemistry comparison between data collected in June 1992, 1998 and 1999 in Caland pit-lake, Ontario. The 1992 data was collected by OMOE. Represented depths are: epilimnion (▪), thermocline (•), and hypolimnion (▲). Concentrations are in $mg \cdot l^{-1}$ except for alkalinity ($mgCaCO_3 \cdot l^{-1}$) and conductivity ($uS \cdot cm^{-1}$).

in succeeding years (Figure 1.7). Since 1992, conductivity, hardness, Ca, Mg, K, Na, Cl, NO_3, TDS, and TKN decreased in concentration in the mixolimnion (Figure 1.7, Appendix C). The pH decreased at each depth since 1992. Alkalinity did not significantly vary between 1992, 1998, and 1999. Sulfate, did not vary significantly between years, but it decreased in the shallower waters. Total-P decreased from 1992 to 1998 but increased again in 1999 (Figure 1.7). With the exception of pH, the hypolimnion concentrations for each of the prior mentioned parameters remained fairly constant.

All of the parameter concentrations varied with depth with the exception of TKN (Figure 1.7, Appendix C). Since 1992 the dissolved oxygen severely decreased (Figure 1.3). In July 1992 there was greater than 4 $mg{\cdot}l^{-1}$ D.O. at 45 m. The 1998 and 1999 D.O. concentrations were less than 1 $mg{\cdot}l^{-1}$ below 23 m depth (Figure 1.3).

1.4 Discussion

1.4.1 Caland pit-lake water chemistry

Caland pit-lake was meromictic with a defined chemocline. The pit-lake could be characterized as hardwater, alkaline lake with a relatively high salt concentration indicated by its specific electrical conductivity and TDS values averaging 650 uS·cm^{-1} and 470 mg·l^{-1} in the mixolimnion to 1090 uS·cm^{-1} and 760 mg·l^{-1} in the monimolimnion, respectively (Table 1.2, 1.3). Cations were dominated by Ca/Mg and the major anion was SO_4 (Table 1.2, 1.3). Pit-lakes with comparable water quality are located in Northern Minnesota's Mesabi Iron Range. The Mesabi pit-lakes were characterized as hardwater, alkaline lakes dominated by the cations Ca/Mg, the major anion though was $HCO_3^-$and to a lesser extent SO_4 (Axler et al., 1992). Water quality in the Caland pit-lake mixolimnion was generally within Provincial Water Quality Objectives (PWQO). *Daphnia* in the shallower waters confirmed the relatively good mixolimnion water quality. With the combination of limestone and dolostone plus more importantly the negligible pyritic waste rock, the impact of the recently scoured rock faces on Caland's pit-lake water chemistry was minimal. However, the operation of the fish farm in the pit-lake clearly influenced the water quality.

1.4.2 Variation between 1998 and 1999

Several parameters decreased in concentration at various depths in 1999 including alkalinity, NO_3, D.O. K, hardness, Mg, Na, SO_4, conductivity, Cl, Ca, DIC, DOC, and TDS (Table 1.2, 1.3, Figure 1.3, 1.4, Appendix B). For several of these ions, increased rainfall in 1999 (Table 1.6) diluted the mixolimnion. Lower concentrations in the upper waters were also likely associated with the sorption of elements on fecal matter from the fish farm and then deposited. Sorption onto surfaces may be a major factor affecting dissolved metal and salt concentrations in the pit lake water column. Under oxidizing conditions at neutral pH, concentrations of positively charged trace metal species are controlled by complexing with and/or sorption on substrates such as clays; organic matter; and aluminum, manganese, and iron-oxides (Miller et al, 1996, OMOE, 1992). There was more organic matter in 1999 due to increased fish density thereby decreasing concentrations of many parameters in the mixolimnion. The decreased concentrations in the upper waters has led to the eventual decrease in Cl, NO_3, SO_4, Mg, and Ca in the monimolimnion. Dissolved oxygen also decreased in 1999 in the shallower water (Figure 1.3). The increased biological oxygen demand of the fish was due to the eight fold increase in fish production (from 20,000 kg early in 1998 to 163,325 kg by the end of 1999) and consequent increase in waste products such as uneaten food, feces, and urine.

A few parameters also increased in concentration in 1999. Reactive $SiO_2$and B increased in the shallower waters in 1999, perhaps due to rising surface levels that contacted and leached feldspar crystals from the rock-wall tonalite gneiss (granite). These elements are distributed in rocks as aluminosilicates (Boyd and Walley, 1972). Total-P and NH_4 increases could be related to the fish farm expansion. Increased waste from feed, feces, and urea would be converted chemically and/or mircrobiologically to ammonia and phosphate (Ackefors et al., 1994). Manganese increased in concentration only off-bottom (Table 1.2, 1.3, Appendix B). Increasing monimolimnion anoxia and decreasing redox potential, especially at the sediment/water interface, would cause the release of Mn from the sediment into the overlying water (Stumm and Morgan, 1981).

1.4.3 Seasonal variations

Time stratified sampling identified variation for most of the parameters (Table 1.2, 1.3, Figure 1.4, Appendix B) with the most obvious effect being the onset of winter. The formation of ice caused the purer water to freeze first and the shallower waters became more concentrated (Figure 1.4, Appendix B). In the ice-free area above the cages, the frigid waters allowed for increased solubility of D.O. thereby replenishing the mixolimnion content (Figure 1.3). This replenishment may not have occurred if ice covered the entire pit-lake. pH decreased due to the annual autumn reduction in photosynthesis, die-off of phytoplankton, and snow being slightly more acidic than rain (Wetzel, 1983). DOC also showed several peaks throughout the year which coincided with heavy rainfalls. DOC concentrations in the drainage waters were relatively similar to pit-lake concentrations (Table 1.2, 1.3, 1.4, 1.5). NH_4 and Total-P also varied with month sampled. These variations could be related to new shipments of trout fingerlings, fish feeding, or fish processing coinciding with sampling time.

1.4.4 Depth variations

Variations with depth were significant for every parameter except Fe (Table 1.2, 1.3). The Pearson correlation hierarchal cluster analysis grouped the variables into three clusters in 1998 and two clusters in 1999 (Figure 1.5). The first cluster for both years represented water chemistry parameters typical of a meromictic lake. The second cluster for 1998 represented parameters that increased in the mixolimnion. The third cluster for 1998 (and second cluster for 1999) consisted of parameters with either no discernable variation with depth or those with higher concentrations in the mixolimnion compared to the monimolimnion (Figure 1.4, 1.5, Appendix B).

1.4.4.1 Cluster 1 - Meromixis

The first cluster consisted of parameters influenced by the development of the meromictic condition. These parameters included hardness, conductivity, alkalinity, Mg, Ca, Na, TDS, Sr, Cl, SO_4, Fe, SiO_2, Mn, DIC, NO_3, and K, and occurred in higher concentration in the monimolimnion compared to the mixolimnion (Table 1.2, 1.3, Figure 1.4, 1.5, Appendix B). The development of meromixis was an identified concern regarding the use of mine pit-lakes for inland fish farming (Axler et al., 1992). Typical meromixis are usually associated with the influx of saline groundwater, influx of saltwater in coastal regions, or surficial freshwater runoff onto a saline lake (Jellison and Melack, 1993). Meromixis may also be caused by the accumulation of solutes derived from the decomposition of organic matter, eg, Wetzel 1983). Snow Lake Fish Farm's associated inputs of organic matter as well as morphology most likely accentuated the meromix in Caland pit-lake. Walker and Likens (1975) also suggested that the establishment of a higher-density monimolimnion may be biologically induced.

1.4.4.2 Cluster 2 - Total phosphorus, ammonium, and total kjeldahl nitrogen

1.4.4.2.i Total phosphorus

The expansion of Snow Lake Fish Farm rainbow trout production from 1998 to 1999 significantly influenced the Total-P concentrations. Total-P in 1998 was generally <MDL (0.005 $mg \cdot l^{-1}$) but increased at all depths in 1999 especially off-bottom (avg 0.02 $mg \cdot l^{-1}$) and at the thermocline (Figure 1.4). 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 $mg \cdot l^{-1}$ in 1988 to 0.094 $mg \cdot l^{-1}$ in 1992 and the Sherman pit total phosphorus levels rose from <0.01 $mg \cdot l^{-1}$ before aquaculture began to levels above 0.25 $mg \cdot l^{-1}$ towards the end of operation in 1995 (Axler et al., 1996). At the LaCloche site, in the North Channel of Lake Huron, P levels rose from <0.005 $mg \cdot l^{-1}$ to an averaged 0.040 $mg \cdot l^{-1}$ in October 1997 (Gale, 1999). Although Caland's Total-P levels increased, they were still below Ontario Ministry of Environment and Energy aquaculture design and operating guidelines of 0.050 $mg \cdot l^{-1}$ and 0.10 $mg \cdot l^{-1}$, respectively, but above the Provincial Water Quality Objectives (PWQO) of 0.010 $mg \cdot l^{-1}$.

Increased Total-P in the thermocline (Figure 1.4) was likely related to the fish food. Enrichment comes primarily from uneaten food and fish waste. Rosenthal et al (1988) showed that approximately 1-30% of dry pellets fed go uneaten, and 25-30% of the food consumed is egested in feces (NCC 1990). Another possibility for the increased Total-P in shallower waters was resuspension. Generally, under oxidized conditions, P precipitates with FeOH complexes, but under anoxic conditions the P is released as Fe^{+++} is converted to Fe^{++} (Mortimer, 1941). This release of phosphorus may have occurred at the 23 m boundary allowing resuspension to the surface waters.

The combination of increasing hypolimnetic anoxia and high P loadings resulted in the increased Total-P release from the sediments. As per Ackefors and Enell (1990) the phosphorus load from net pen aquaculture is about 80% solid and settles relatively quickly to the bottom.

1.4.4.2.ii Total kjeldahl nitrogen and free ammonium

TKN doubled in 1999 to 0.4 $mg \cdot l^{-1}$ and consequently, organic-N also doubled to approximately 0.3 $mg \cdot l^{-1}$. These increases were each affiliated with the increased fish production. In Minnesota, Axler et al (1996) found that nitrogen levels (NO_3+NO_2+NH_4) rose from 0.363 $mg \cdot l^{-1}$ in 1988 to 2.043 $mg \cdot l^{-1}$ in 1993 in response to intensive net pen aquaculture. Feeding fish held captive in cage pens causes a compounding problem of waste production both with food and feces. Bergheim et al., (1991) stated that only 25-30% of the nitrogen and phosphorus in feed is typically retained in the fish, 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 (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 BOD), and potentially to un-ionized ammonia (NH_3) toxicity (Axler et al., 1996).

In Caland pit-lake, the 1998 NH_4 levels were generally <0.06 $mg \cdot l^{-1}$ but in 1999 increased in the shallower waters to 0.15 $mg \cdot l^{-1}$ (Table 1.2, 1.3, Appendix B). 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 7.5 is 0.859 whereas at pH 8.0 is 2.67 (Ackefors et al.,1994). In Caland pit-lake at an average summer temperature of 20 °C, epilimnion pH nearing 8.0, and average ammonia level of 0.15 $mg \cdot l^{-1}$, the un-ionized (toxic) ammonia is 0.0057 $mg \cdot l^{-1}$. However, this concentration is still much less than the un-ionized PWQO of 0.02 $mg \cdot l^{-1}$ (OMOEE, 1994).

In 1999, the NO_3-N levels decreased at each depth in Caland pit-lake (Table 1.2, 1.3). This decrease was likely from denitrification caused by increasing anoxia in the monimolimnion. Axler et al., (1995) suggested using denitrification as a management tool to provide some control of NO_3 buildup in Minnesota's extensive fish farms.

1.4.4.3 Cluster 3 - DOC and B, and pH, temperature and D.O.

Parameters in this cluster showed either no discernable variation with depth (DOC and B) or had higher values in the mixolimnion (pH, temperature, and D.O.) compared to the monimolimnion.

1.4.4.3.i Dissolved organic carbon & boron

DOC and B were seemingly similarly influenced by rainfall and inflow. Boron concentrations in 1998 were relatively low (avg MDL at 0.05 $mg \cdot l^{-1}$) , yet increased in all measured depths in 1999 (Table 1.2, 1.3). Boron is found in rocks as it is contained in some aluminosilicate minerals (Boyd and Walley, 1972), albeit not in large quantities. The increased rainfall and drainage inflow with subsequent rock weathering influenced its levels such that epilimnetic B averaged 0.128 $mg \cdot l^{-1}$ (Table 1.3).

DOC levels in Caland averaged 3.3 $mg \cdot l^{-1}$. Particulate organic carbon (POC) levels were not analysed but generally occur in a DOC:POC ratio of 10:1, thereby reflecting no significant concentration difference in total organic carbon (TOC). Comparably, in Minnesota there was no significant buildup of TOC, concentrations were roughly 3 $mgC \cdot l^{-1}$ (Axler et al., 1996). A feature of DOC is the absence of strong vertical stratification or seasonal fluctuations, despite the parallel stratification and seasonal pulses of metabolic activity (Wetzel,1983). Highest concentrations occurred during summer stratification in the epilimnion. DOC also showed several peaks (Appendix B) which coincided with heavy rainfalls, and therefore may have been influenced by terrestrial humic substances in the inflow drainage. The most common precursors are humic substances derived from the decomposition of plant and animal tissue which constitute at least half of the DOC in natural waters (Wetzel et al., 1972). The fish farm excretions would also add to the DOC pool, from bacterial chemosynthesis of organic matter.

1.4.4.3.ii Temperature, pH, & dissolved oxygen

Caland's mixolimnion temperature profile was consistent with temperate lakes. It thermally stratified in the summer with a defined thermocline, and destratified with spring and fall turnovers. The ice in/out period (mid December and early May, respectively) coincided with other lakes in the area. Ice remained open above the fish pens, which influenced the water temperature as inverse stratification was observed to a depth >6 m. In the monimolimnion, temperatures started to increase circa 35 m depth (Table 1.2, 1.3, Figure 1.3). This increase was indicative of meromictic environments, having high input and decay of organic matter.

The pH steadily decreased with increasing depth. In the epilimnion pH averaged 7.86 versus off-bottom level of 7.01 (Table 1.2 and 1.3). Decay of organic matter likely lowered the pH. Respiratory generation of CO_2 and the decomposition process decreases pH (Wetzel, 1983).

Caland's D.O. profile (Figure 1.3) was indicative of a meromictic environment with an oxygen depleted monimolimnion. An earlier paper by McNaughton et al., (1999) suggested that the monimolimnion oxygen depletion may have been caused by the oxidation of H_2S which occurs spontaneously in the presence of oxygen (Jorgensen et al., 1979). In Caland, although sulfate concentrations below the chemocline were relatively high ($>400\ mg \cdot l^{-1}$), sulfide concentrations were $<0.1\ mg \cdot l^{-1}$.

In the mixolimnion, the August 1999 D.O. profile had a distinct negative heterograde shape (Figure 1.3). This oxygen consumption paralleled the farm expansion with the combined increase in fish respiration and organic decay.

As previously stated, the oxygen depletions also occurred directly below the deepest reaching fish pens at a depth of 20 m (Figure 1.3). Between 1998 and 1999, dissolved oxygen was consumed almost equivalent to freshwater inflow, totaling approximately $5x10^6$ m^3. Organic matter trapped at the bottom of the cages was mainly responsible for biological oxygen demand at this depth. Organic accumulation in the bottom of the nets totaled approximately one tonne/year which eventually is released to the bottom strata (Lindsay, pers comm. 2000). 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 20 m (Figure 1.3).

1.4.5 Influence of drainage or climate

Generally, effluent affected by acid mine drainage (AMD) will show the following characteristics: a pH of between 2 to 4 associated with sulfuric acid generation, concentrations of dissolved iron typically in the range of 100 - 3000 $mg \cdot l^{-1}$, elevated (1 - 200 $mg \cdot l^{-1}$) concentrations of other dissolved metals such as copper, zinc, and lead, and high concentrations of dissolved salts including calcium, magnesium, aluminum, and sulfate (100 - 30 000 $mg \cdot l^{-1}$) (Ritchie, 1994). At Caland, the drainage pH was above 7. The south drainage exhibited dissolved salts with concentrations greater than 100.0 $mg \cdot l^{-1}$ for SO_4, Ca, and Mg (Table 1.4, 1.5, Figure 1.6). The west drainage had elevated SO_4 levels.

Between 1998 and 1999 the drainage water chemistry was relatively similar (Table 1.4, 1.5, Figure 1.6). Drainage, as previously mentioned, did influence pit-lake water chemistry since B and DOC concentrations in the drainage water were relatively similar to concentrations in the pit-lake. Also, drainage water may have influenced the mixolimnion since several of the parameters were less concentrated and therefore would mix and dilute the pit-lake mixolimnion. Furthermore, the quantity of overland inflow was significant enough to raise surface water level approximately 5 m over the two-year study period.

Although, the 1998 climate data (Table 1.6) reflected low rainfall along with slightly warmer summer temperatures, this did not noticeably influence the pit-lake water chemistry since surface inflow was still substantial. Increased rainfall though in 1999 did dilute concentrations in the shallower waters.

1.4.6 Influence of fish farm since 1992

The majority of the measured parameters (conductivity, hardness, Ca, Mg, K, Na, Cl, NO_3, TDS, and TKN) in the mixolimnion decreased in concentration between 1992 and 1998 (Figure 7). Lowered concentrations in the upper waters were likely associated with the sorption of elements on fecal matter from the fish farm, and precipitated.

Total-P, however, decreased from 1992 to 1998 but increased again in 1999 (Figure 1.7). In 1992, Total-P was approximately 0.01 $mg \cdot l^{-1}$ which was associated with the density and feeding of Chinook (91,000 kg). Since that time a combination of both freshwater inflow diluting the nutrient enrichment, and the inability of Snow Lake Fish Farm to increase or maintain production, Total-P levels decreased to <0.005 $mg \cdot l^{-1}$ by early 1998. Similarly, following the closure of Minnesota's net pen activities phosphorus levels decreased from 0.094 $mg \cdot l^{-1}$ in 1992 to near 0.010 $mg \cdot l^{-1}$ in 1994 (Axler et al., 1996).

Since 1992 D.O. has severely decreased. In July 1992 >4 $mg \cdot l^{-1}$l D.O. occurred at 45 m, but by 1998 concentrations were <1 $mg \cdot l^{-1}$ below 23 m depth (Figure 1.3). This oxygen depletion in the monimolimnion occurred prior to the 1998 Snow Lake Fish Farm expansion.

Between 1992 and 1998, with low fish production and approximately 40 m of freshwater inflow, >60 m of oxygen were consumed (ie. 40 m inflow + >20 m oxic water below fish pens in 1992, totaling approximately 6×10^7 m^3). In Caland pit-lake, dead fry and fish, feces, wasted feed, and the processed fish guts and trimmings, were deposited into the water column. Axler et al.(1996) stated that organic enrichment from uneaten food and fish feces may lead to hypolimnetic and sediment anoxia. This condition has been realized at the LaCloche cage fish farm, in the North Channel of Lake Huron. This operation developed D.O. levels from the surface to 12 m depth of only 5-9 $mg \cdot l^{-1}$, and 0 $mg \cdot l^{-1}$ at depths from

13 m to 41 m (Gale, 1999). These conditions encompassed the entire hypolimnetic volume, over a 250 ha area.

1.4.7 Management

The Habitat Advisory Board of the Great Lakes Fishery Commission and the Great Lakes Water Quality Board of the International Joint Commission (1999) suggested that mine pits are not suitable sites for inland fish farming. Their Roundtable discussions, Addressing Concerns for Water Quality Impacts from Large-scale Great Lakes Aquaculture, insisted that pit-lakes are unnatural, very deep, have little vegetation, no littoral zone, and low nutrients. In pit-lake environments the input of nutrients from feeding and fish waste results in a more intensive and immediate problem than other more natural sites. Contrary, though, to both the Roundtable concerns and to the findings at Snow Lake Fish Farm, we continue to believe that abandoned open pit mines are prime locations for inland fish farming. These contained environments ensure no mixing with wild stocks, and no influence of anti-disease agents affecting wild stocks or other aquatic organisms. Furthermore, increasing water levels do alleviate nutrient increases and degraded pit-lakes appear to recover relatively quickly.

There are some major concerns such as dissolved oxygen consumption and hygiene, but these concerns can be remedied. Mixolimnion dissolved oxygen can be enhanced through the implementation of aeration systems such as windmills, subpumps, or fountains. Waste disposal at the fish farm can be limited either through the use of diapers, plastic bags, and simply depositing materials at a secluded area on shore.

Mining companies have suggested (Mike Sudbury, Falconbridge Environment, pers. comm, 1998) that one alternative for remediating abandoned open pit mines may be achieved through implementation and promotion of fish farming. Utilizing these pits for aquaculture would decrease mining closure costs and promote small business. If mining companies intend to use fish farming as an alternative method for rehabilitation they must take more responsibility towards research and development of these farms. Inland aquaculture in pit-lakes requires further research on oxygen regeneration, limit of waste build-up, proper waste disposal, accurate determination of carrying capacity through nutrient enrichment and fish density, and water quality recovery. The mine company and fish farmer must also realize that eventual outflow may be detrimental to the receiving water body. Research on the use of filters such as layers of gravel and/or wetlands placed at possible future outlets should be considered.

2. Water quality and aquacultural potential of Hogarth pit-lake, located near Atikokan, Ontario

2.1 Introduction

Research on open pit lakes has recently increased to develop alternatives for post-mining environmental remediation (Stevens and Lawrence, 1998; Doyle and Runnells, 1997). Associated with their remediation are water quality concerns, since, at some point, this water will either outflow, become part of the groundwater table, be used for recreation or inland fish farming, or act as sites for deposition of waste materials.

The chemical composition of the receiving pit-lake waters primarily depends on two processes (1) chemical weathering reactions that release solutes into inflowing waters, and (2) subsequent evaporative concentration leading to the precipitation of minerals that affects the composition of the remaining waters (Eugster and Hardie, 1978). Generally with mining activities, acid mine drainage (AMD) occurs from tailings produced during the mineral extraction process or from waste rock piles (Gould et al., 1994). This acidic drainage is the most effective agent for weathering of the carbonates and silicates, which constitute the bulk of the surface rocks. AMD derives its acidity by oxidation of sulfides to sulfates in conjunction with precipitation of metal oxides or hydroxides: $4FeS_{2(s)} + 15O_{2(g)} + 8H_2O_{(l)} \rightarrow 2Fe_2O_{3(s)} + 8SO^{2-}_{4(aq)} + 16H^+_{(aq)}$ (Kuyucak, 1999). AMD is characterized by a pH of 4 or less, high sulfate content, elevated levels of dissolved metals such as iron and aluminum, and high concentrations of dissolved salts. The oxidation of sulfides such as pyrite and pyrrhotite in tailings ponds, responsible for the acidic drainage generated at these sites, can be caused both by chemical and biological reactions. Microbial activities that catalyze AMD are also associated with the bioleaching of metals (Brierley, 1978; Lundgren and Silver, 1980). Rates of weathering and leaching are greatly accelerated with unlimited accessibility of oxygen. Sufficient drainage from these active sites will influence receiving pit-lake waters due to the increased solute loadings. In some instances, the increased weathering reactions may increase the total dissolved solids sufficiently to create saline conditions.

Saline lakes vary from those with almost freshwater, in dry temperate climates, to viscous organic soups covered with seasonal crusts of salt, such as those in tropical east Africa. Generally, lakes containing more than 3ppt salinity are considered salt lakes (Williams, 1978). Another important feature which distinguishes salt from freshwater lakes is that salt lakes are much more sensitive to climatic change (Williams, 1996). This enhanced sensitivity reflects the nature of their existence. Salt lakes are the product of a delicate balance between basin inputs (precipitation over the lake plus inflows from rivers) and outputs (evaporation and seepage outflows). When any of the four elements of this equation are even slightly disturbed, significant hydrological and hence environmental, biological, and other changes occur in the lake (Williams, 1996).

The relative proportions of the major solutes vary greatly from one saline lake to the next and are controlled by rock dominance, atmospheric precipitation, and the evaporation-precipitation process (Gibbs, 1970; Feth, 1971; Kilham, 1975; Stallard, 1980; Stallard and

Edmond, 1981). Rock-dominated waters are rich in calcium and bicarbonate ions, and are more or less in equilibrium with materials of their drainage basins. Positions within this grouping depend on the climate, basin relief, and the composition of rock material in the basin. Precipitation, as a mechanism, generally is associated with coastal lakes where Na and Cl are carried by wind and water droplets, and deposited inland. Most natural saline waters are dominated by Na and Cl. The third mechanism influencing salinity of surface waters is evaporation and fractional crystallization with subsequent sedimentation of mineral salts (Wetzel, 1983). In open-pit lakes, pit-lake water chemistry is predominantly determined by proximal rock-dominance but the evaporation-precipitation process also plays an influential role (Miller et al., 1996; Williams, 1996).

Pit-lake waters range in quality from very polluted to sometimes serving as a drinking water supply in nearby communities. As concurred by Banks et al (1997), mine waters need not be perceived as problems, but can often be regarded as sources of drinking water or for industrial uses such as sewage treatment, tanning, and industrial metals extraction. In addition some abandoned mine sites may serve as sites for inland aquaculture. This appeals to mining companies as a less costly alternative method for post operation remediation. In this portion of the study, the water quality of a pit-lake, Hogarth, known to be high in dissolved solid levels was examined. The objectives were to determine i) if water quality varied with depth and with season; ii) the likely mechanisms influencing the water quality; and iii) if these conditions would support aquaculture for post-mining activities.

2.2 Method

2.2.1 Study Site

The study site was Hogarth pit-lake, an open pit located at the former site of Steep Rock Mines at Steep Rock Lake (48° 48'N, 91°39'W) near Atikokan, Ontario, Canada. Hogarth pit-lake, is located in the northern part of the former Middle Arm of Steep Rock Lake (Figure 1.1). It has a depth of 160 m with a surface area of approximately 100 hectares. The pit-lake continues to increase in depth from rainfall, runoff, and groundwater seepage, and, by the year 2030, Hogarth pit-lake will be connected to Caland pit-lake, located in the former East Arm of Steep Rock Lake. At that time, the maximum depth of the newly connected pits will be 425m with a combined area of 13.4 km^2(Chapman, 1997). Currently, Hogarth pit-lake has a ladle shaped morphology with the cup of the ladle being a bowl/cone shaped basin, with a shallower handle extending southward (Figure 2.1). Hogarth pit-lake has a very low surface area to depth ratio. This lake is steep sided, well-sheltered by high rock walls, and has a drainage basin averaging 5.1 km^2.

Hogarth pit-lake is located at the southern margin of the granite-greenstone Wabigoon Subprovince of the Superior Province of the Canadian Shield (Ontario Ministry of Northern Development and Mines, 1994). 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 bodies. 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

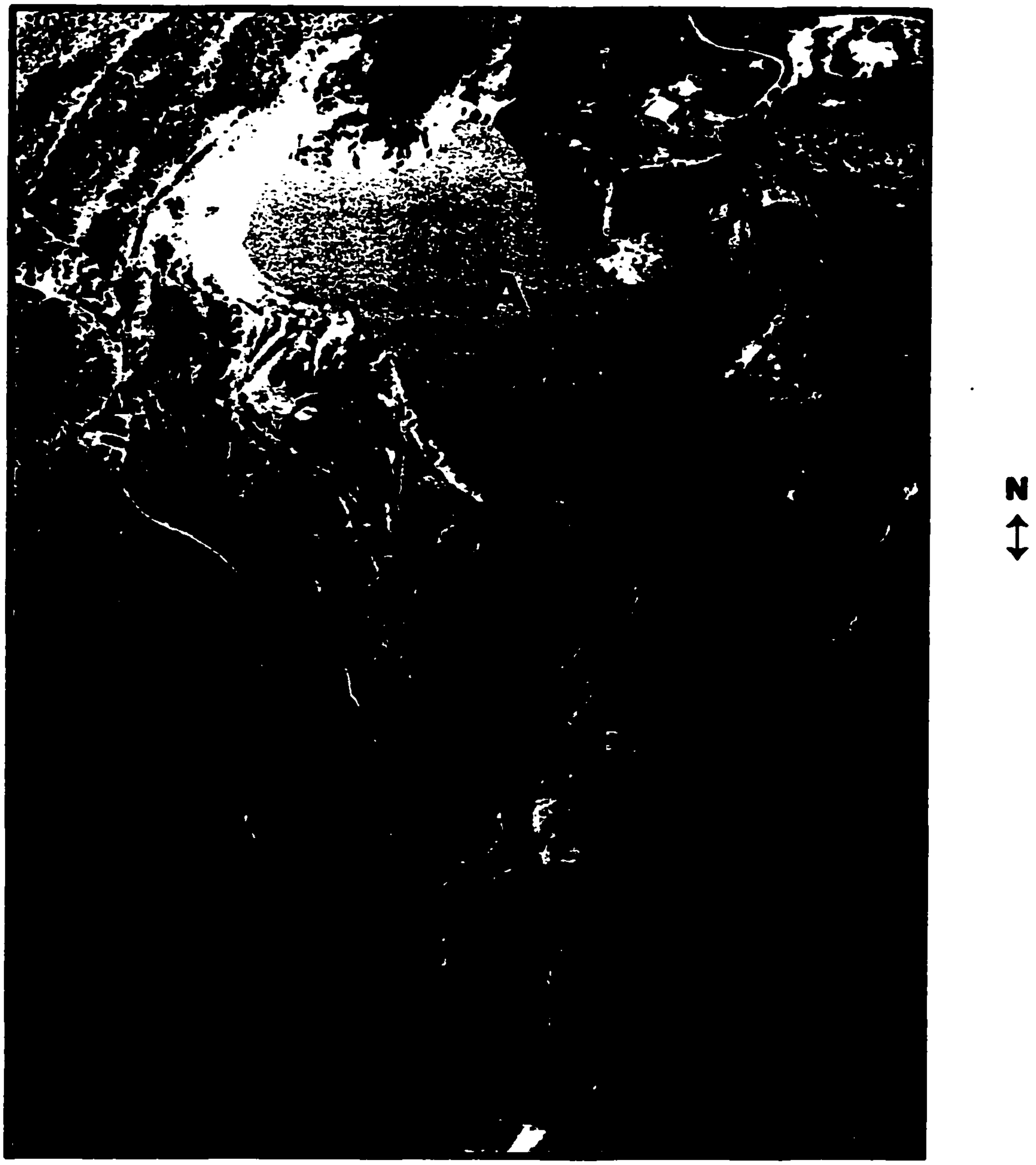

Figure 2.1: Site location map. Sampling occurred at two water chemistry stations in Hogarth pit-lake, Ontario. Station A was 160m deep and B was 30m deep. Four drainages were also sampled.

Carbonate Formation which varied 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 760m, width of 50m, and length of over 4km. 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 give credence to this hypothesis.

The occurrence of iron bearing rocks in the Atikokan region was first recognized by Smyth in 1891, but it was not until 1930 that the ore bodies were located below Steep Rock Lake (Steep Rock Mines, 1943). Since the iron ore body was located under Steep Rock Lake, the lake had to be drained. This was accomplished by diverting the inlet, the Seine River, to flow through Finlayson Lake and installing a series of dams (Figure 1.1).This diversion was completed in 1943, and the net effect moved the Seine River two miles westward for eight miles of its course. The Middle and East Arms of the lake were pumped dry and about 205 million m^3 of overburden were removed. Steep Rock Iron Mines mined the

Middle Arm of Steep Rock Lake (Hogarth, Errington, and Roberts Pits) (Pye, 1968). The Hogarth mine opened in 1953. From 1954 to 1977, the life of the Hogarth Mine, tons mined totaled 25,606,184. After mining ceased, the rights to the land reverted to the Government of Ontario and the pit began to naturally fill with water.

2.2.2 Field Procedures

Two sampling stations were selected in Hogarth pit-lake: one in the cup of the basin (160 m) and one two/thirds down the handle (30 m) (Figure 2.1). Water was sampled at each of the stations in the epilimnion (1 m), thermocline, hypolimnion (30 m) and 1 m off bottom. Water was collected and sampled six times in 1998, commencing monthly from June to September, then before ice cover (early November) and through the ice (late February, 1999). The second year of sampling commenced after ice-out in mid May and continued monthly from June to September. Sampling also occurred in November and through the ice in March of 2000 for a total of seven trips for 1999. Lake water was collected at each station using a Kemmerer Bottle lowered with calibrated rope. Secchi depth was measured using a secchi disk with calibrated line. Time of day, air temperature, and cloud cover were recorded, and wind speed estimated. Dissolved oxygen and temperature profiles were obtained from each station to the cable depth of 50 meters using Yellow Springs Instrument (YSI) Model #51B. Estimates of off-bottom D.O. and temperature were obtained by placing the YSI probe into the contained water sample in the Kemmerer bottle, limiting exposure to atmospheric influences. Drainage water was also collected during the ice-free periods totaling eleven sampling trips, from stations located on the north, northeast, east and southeastern walls. No noticeable drainages were detected on the south or west walls. Drainage water was collected by surface grab method in a polyethylene bottle.

Each water sample was divided into three separate volumes before being transported back to the Lakehead University Environmental Lab (LUEL) for analysis. Each sample was divided into 3 separate volumes: 250 ml was preserved with 1.0 ml of HNO_3 (conc) for total metal determinations; 250 ml was preserved with 1.0 ml H_2SO_4 for total kjeldahl nitrogen; total phosphorus, and free ammonium; and 1 L was unpreserved for total suspended and total dissolved solids, alkalinity, conductivity, dissolved organic and inorganic carbons, silicates, cations, and anions. Field sampling quality controls included 1 field blank, 1 travel blank, and 1 duplicate sample taken each sampling trip. Water from each sample was cooled on ice prior to shipment back to LUEL. Table 1.1 lists the physical and chemical factors examined at each sampling station.

2.2.3 Analytical Procedures

Analytical methods followed the procedures outlined in Chapter 1. The following tests were also conducted.

Arsenic concentrations were tested in June 1999 by Accurassay Laboratories located in Thunder Bay, Ontario, three days after sampling. The method for analysis was Hydrides by Atomic Absorption Spectrometry with a method detection limit (MDL) <0.002 (Greenberg et al., 1992)

In May 1999 a toxicity LC_{50} was conducted on *Daphnia magna* by the Aquatic Toxicity Research Center (ATRC) located at Lakehead University, Thunder Bay, Ontario. A 1 L composite sample of pit-lake water was used for this test following the procedure outlined in Environmental Protection Services (1996).

2.2.4 Data Analysis

A multivariate computer program from the Statistical Package for Social Sciences (SPSS) was used for the data analysis. The water chemistry (pit-water and drainage) was summarized using means and standard deviation. To test for seasonal trends, a Repeated Measures Anova was used for each of the 25 water chemistry parameters (parameters that were below method detection limits for >50% of the measures were removed from the data set). Data collected from each station was entered into the program. Repeated Measures Anova also allowed testing for significant ($p<0.05$) differences between the two sampling years, as well as with depth. Since this program required a balanced input, data collected in May 1999 was eliminated from the set.

2.3 Results

2.3.1 Hogarth pit-lake water quality

The mean and standard deviation for 25 water chemistry variables, separated by depth, with six replicates for 1998 and seven replicates for 1999 are shown in Table 2.1 and 2.2, respectively. Hogarth pit-lake was milky-olive in color, and upon closer visual examination had notable iron floc. The secchi depth average was 1.5 m. The water in the pit was neutral with a combined depth average pH of 6.88. The average pH in the epilimnion was 7.35 and decreased with depth where the average off-bottom pH was 6.73 (Table 2.1, 2.2). The pit-lake was buffered with average alkalinity of 94.27 $mgCaCO_3 \cdot l^{-1}$ and average hardness of 1681 $mgCaCO_3 \cdot l^{-1}$ (calc) (Table 2.1, 2.2). The conductivity and dissolved solids data reflected a semi-saline pit-lake with potential for meromixis upon freshwater inflow. Several of the water chemistry parameters (SO_4, Ca, Mg, Ni, and TDS) were elevated in concentration compared to natural lakes (Table 2.1, 2.2). The dissolved oxygen profile resembled those of an oligotrophic environment (Figure 2.2), with no oxygen minima with depth.

The shallower water in Hogarth pit-lake thermally stratified during the summer (Figure 2.2) and destratified partially in the fall. This entire water column did not overturn as it is well protected by high rock walls, which diffuses much of the wind energy required for the process.

Table 2.1: The 1998 water chemistry means and standard deviations for Hogarth pit-lake. Concentrations are in mg·l^{-1} except for alkalinity (mgCaCO$_3$·l^{-1}), conductivity (uS·cm^{-1}), hardness (calc.), pH, and temperature (^{0}C). The ^{abc} indicates significant ($p<0.05$) differences between the two years of study (^{a}), with season (^{b}), or with depth (^{c})

	Epilimnion		Thermocline		Hypolimnion		Off-bottom	
	mean	std.dev.	mean	std.dev.	mean	std.dev.	mean	std.dev.
Alkalinity ^{abc}	94.91	2.11	97.73	1.17	96.30	1.38	96.67	1.46
Conductivity ^{abc}	2384.78	158.20	2381.30	162.04	2385.65	168.28	2383.48	170.31
DIC ^{abc}	24.70	4.21	26.49	5.04	25.40	5.60	25.92	6.12
DOC ^{ab}	1.75	1.05	1.63	0.88	1.60	0.87	1.72	1.03
Hardness	1728.44	64.91	1722.38	58.60	1704.27	122.24	1713.11	124.14
Cl^- ^{abc}	17.18	2.06	17.27	2.33	17.52	2.25	17.54	2.29
NO_3^- ^{abc}	1.30	0.23	1.33	0.24	1.39	0.25	1.38	0.25
SO_4^- ^{ab}	1792.13	238.44	1780.15	253.28	1791.08	246.12	1788.44	233.90
Ca ^{abc}	342.56	16.32	341.29	14.79	344.49	13.43	346.62	12.92
K ^{b}	7.03	0.39	6.92	0.36	6.85	0.42	6.97	0.48
Mg ^{bc}	211.96	6.94	211.31	5.65	210.28	5.56	211.12	5.78
Na ^{bc}	25.20	0.77	25.33	0.53	25.21	0.75	25.27	0.69
B ^{b}	0.079	0.06	0.093	0.081	0.095	0.071	0.096	0.069
Fe	0.27	0.13	0.27	0.11	0.21	0.12	0.54	0.49
Mn ^{abc}	0.15	0.04	0.19	0.08	0.05	0.04	0.13	0.08
Ni	0.058	0.014	0.066	0.007	0.069	0.006	0.072	0.008
Sr	1.44	0.04	1.44	0.06	1.45	0.04	1.45	0.05
Zn ^{b}	0.013	0.005	0.013	0.004	0.013	0.004	0.013	0.006
pH ^{bc}	7.43	0.49	6.72	0.28	6.64	0.21	6.67	0.31
SIO_2 ^{abc}	10.11	2.07	10.51	2.08	10.94	2.41	10.82	2.34
TKN ^{ab}	0.056	0.051	0.048	0.033	0.05	0.04	0.067	0.067
TDS ^{abc}	2476.89	137.68	2537.41	253.18	2470.30	88.60	2447.99	113.58
TSS	10.45	1.43	10.34	1.61	10.28	2.31	11.00	1.65
Temp (°C) ^{bc}	14.83	7.91	11.28	5.21	4.02	0.38	4.00	0.39
D.O. ^{bc}	9.50	1.88	9.80	1.55	8.61	0.54	8.36	0.56

Table 2.2: The 1999 water chemistry means and standard deviations for Hogarth pit-lake. Concentrations are in $mg \cdot l^{-1}$ except for alkalinity ($mgCaCO_3 \cdot l^{-1}$), conductivity ($uS \cdot cm^{-1}$), hardness (calc.), pH, and temperature (^0C). The ^{abc} indicates significant ($p<0.05$) differences between the two years of study (^{a}), with season (^{b}), or with depth (^{c}).

	Epilimnion		Thermocline		Hypolimnion		Off-bottom	
	mean	std.dev.	mean	std.dev.	mean	std.dev.	mean	std.dev.
Alkalinity ^{abc}	79.63	11.50	90.02	7.09	97.28	3.25	97.91	3.25
Conductivity ^{abc}	2042.38	75.88	2143.13	44.68	2198.13	32.50	2200.00	32.86
DIC ^{abc}	17.42	2.22	20.63	2.22	22.31	2.11	22.24	2.19
DOC ^{ab}	2.22	1.99	2.20	1.88	2.19	1.93	2.24	2.04
Hardness	1551.59	53.89	1617.08	63.36	1667.91	50.13	1680.26	63.74
Cl^- ^{abc}	13.33	0.76	14.38	0.57	15.15	0.61	15.21	0.55
NO_3^- ^{abc}	0.90	0.06	0.98	0.06	1.03	0.04	1.04	0.04
SO_4^- ^{ab}	1324.36	200.38	1394.38	221.34	1427.81	224.85	1433.21	223.48
Ca ^{abc}	302.96	11.26	315.54	13.48	326.92	12.94	328.53	13.58
K ^{b}	6.03	0.37	6.18	0.40	6.64	0.32	6.68	0.30
Mg ^{bc}	193.08	7.67	201.35	8.66	206.80	6.58	208.82	8.78
Na ^{bc}	22.35	0.87	23.49	1.01	24.48	0.84	24.79	1.12
B ^{b}	0.141	0.064	0.149	0.072	0.131	0.068	0.136	0.067
Fe	0.37	0.13	0.46	0.29	0.20	0.08	0.40	0.40
Mn ^{abc}	0.38	0.05	0.32	0.08	0.13	0.07	0.17	0.10
Ni	0.065	0.006	0.064	0.006	0.061	0.006	0.063	0.006
Sr	1.28	0.10	1.36	0.07	1.38	0.10	1.40	0.09
Zn ^{b}	0.012	0.005	0.012	0.005	0.011	0.002	0.012	0.004
pH ^{bc}	7.28	0.34	6.84	0.23	6.74	0.18	6.80	0.31
SiO_2 ^{abc}	7.03	1.04	7.83	1.16	8.41	1.04	8.32	0.88
TKN ^{ab}	0.119	0.062	0.138	0.078	0.139	0.068	0.131	0.069
TDS ^{abc}	2215.10	113.64	2340.71	108.46	2362.16	84.49	2391.39	89.10
TSS	9.67	1.45	10.05	1.37	9.63	2.13	10.33	2.01
Temp (^oC) ^{bc}	13.82	7.53	10.16	5.68	3.97	0.30	3.99	0.30
D.O. ^{bc}	10.20	1.93	9.83	1.32	9.17	1.14	8.88	1.02

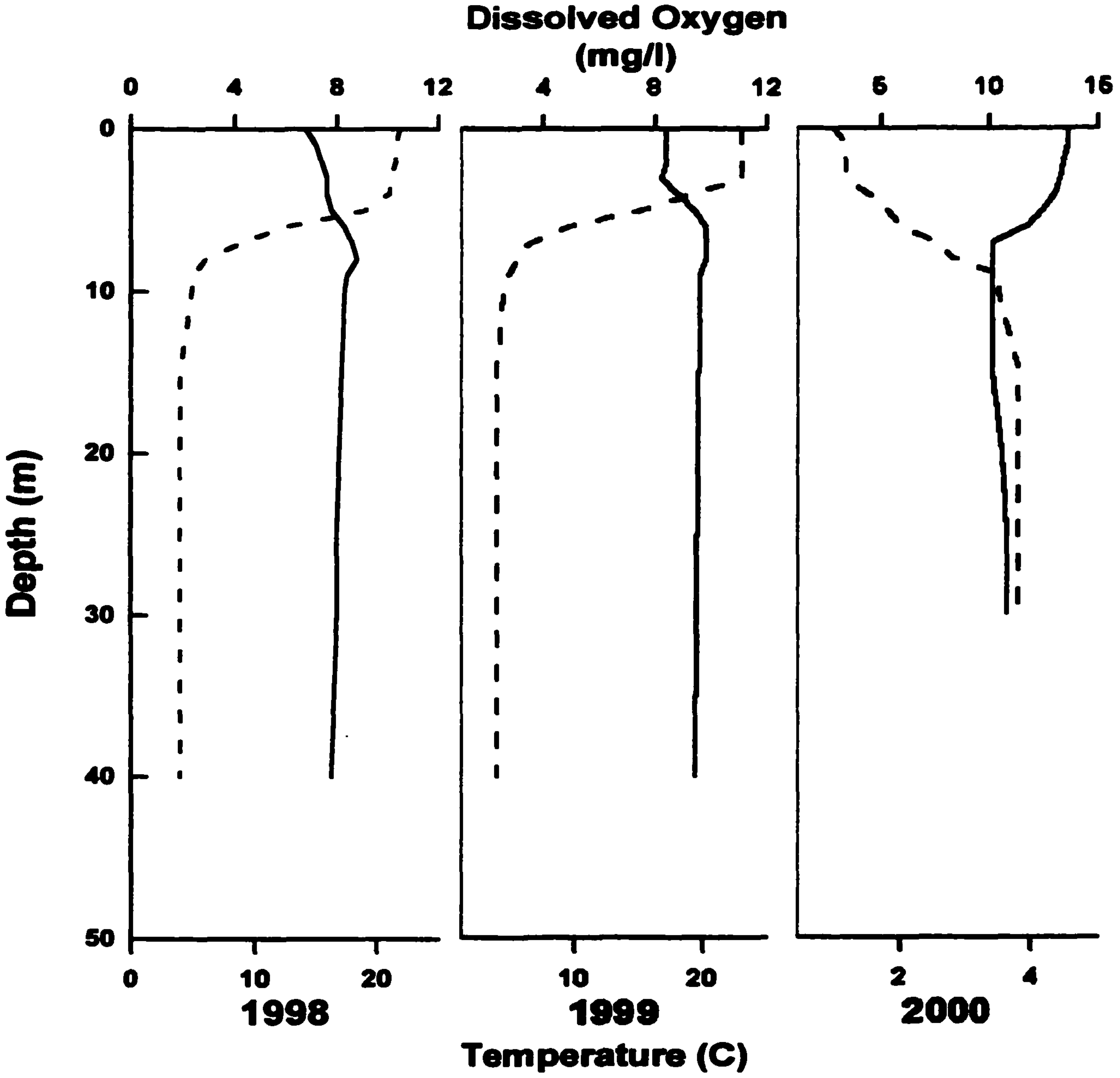

Figure 2.2: Dissolved oxygen (—) and temperature (--) profiles for August 1998 and 1999, and winter 2000. Each profile is an average of the two sampling stations in Hogarth pit-lake, Ontario.

2.3.2 Variation between years, season, or depth

The Repeated Measures Anova showed that alkalinity, conductivity, Cl, NO_3, Ca, Mn, SiO_2, TDS, DOC, DIC, SO_4, and TKN were significantly different between 1998 and 1999 (Table 2.1, 2.2). Figure 2.3 and Appendix D show the changes graphically during the sampling period for each parameter. The 1998 pit-lake water column chemistry appeared to be vertically homogeneous with the exceptions of water temperature and D.O. (Figure 2.2), and Mn and pH (Figure 2.3). Increased variation with depth for several parameters occurred in 1999 and caused the differences between years. In 1999 the epilimnion and thermocline became diluted with rainfall which decreased the concentrations of alkalinity, conductivity, TDS, Ca, DIC, SiO_2, and Cl (Appendix D). SO_4 (Figure 2.3) and NO_3 (Appendix D) in 1999 decreased at each depth. Conversely, TKN (Appendix D) increased at each depth and Mn (Figure 2.3) increased only in the shallower depths.

Most of the parameters that varied between 1998 and 1999 (except DOC, TKN, and SO_4), as well as Mg, Na, pH, water temperature, and D.O., differed significantly with depth (Table 2.1, 2.2). Parameters that varied seasonally included the parameters that varied between the two years, with depth, as well as Zn, K, and B (Table 2.1, 2.2, Figure 2.3, Appendix D). Most of these variables differed primarily because of the onset of winter and ice formation. Hardness, Sr, Fe, Ni, and TSS were not varied in concentration for any of the sources (Table 2.1, 2.2, Appendix D).

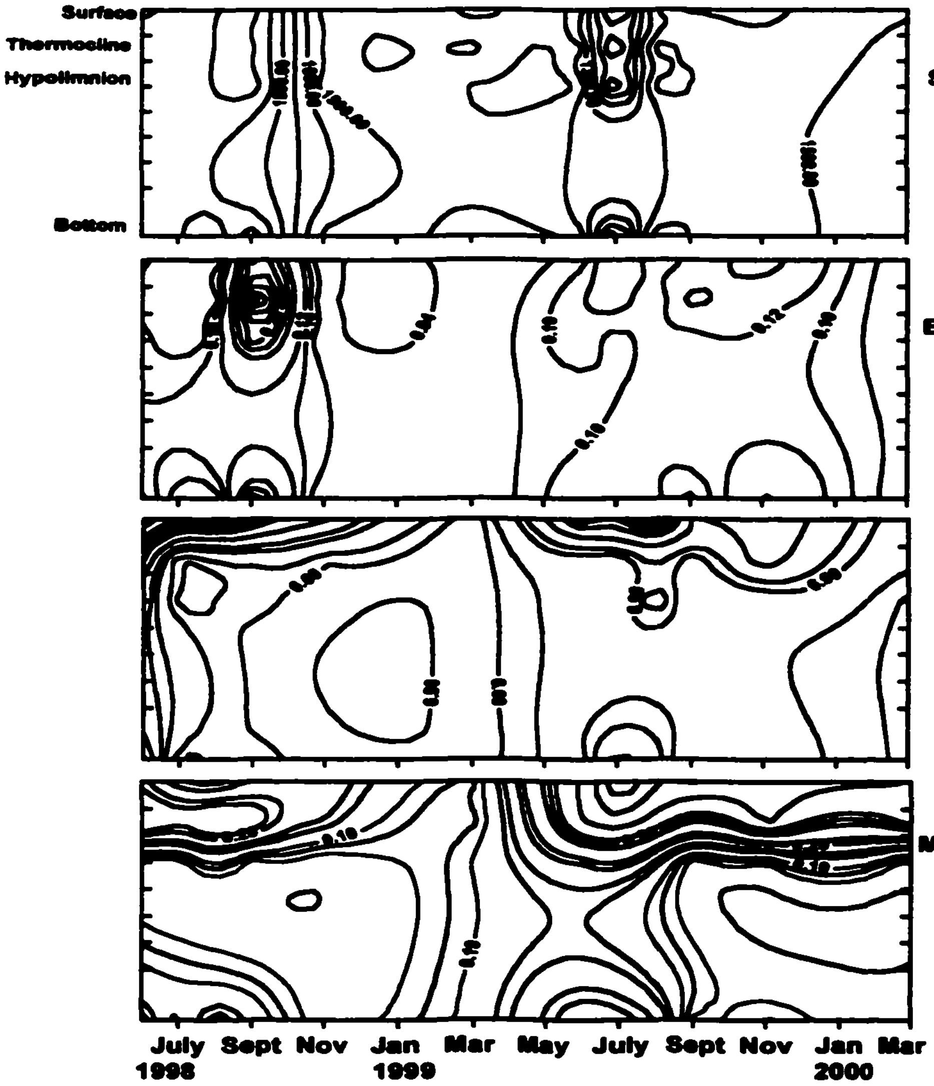

Figure 2.3: Sulfate, boron, pH, and manganese average concentrations of the two sites at represented depths shown for each sampling occasion in Hogarth pit-lake. Each of the parameters are in mg·l^{-1} except pH.

2.3.3 Drainage and climate

Means and standard deviation for 27 water chemistry parameters measured in the four drainages are shown in Table 2.3 and 2.4. The large standard deviations were attributed to variations in rainfall throughout the ice-free sampling period. Figure 2.4 contains typical variations for drainage parameters exhibiting AMD between 1998 and 1999. The northeast drainage was not located prior to the 1999 sampling period. Drainage into the pit varied immensely. pH ranged from 2.71 at the seep to 7.36 at the northeast drainage. SO_4, Ca, Fe, and Mn each increased in concentration with decreases in pH (Figure 2.4). Magnesium, though, had higher levels in the southeast drainage compared to the seep.

A few hundred meters above the southeast drainage, on an east slope, a white gel oozed from the ground (1.5 m^2 area) and leached downslope killing the vegetation prior to entering the southeast drainage. The gel mainly consisted of 60 cps Al and 40 cps O (measured by Energy Dispersive Spectrum). This gel was not noticed until after a rainstorm in late October 1998 and persisted throughout 1999. The gel did not appear to mix with the drainage water but instead coated the rock substrate. Upon entering the pit-lake, the gel settled on the bottom. This substance was believed to be bauxite. A similar gel substance known as ferruginous bauxite was noted during the mining operations at the Errington site (Steep Rock Resources Inc., 1978). Bauxite may have formed in the area during historical tropical conditions, circa 70 Mya or 2900 Mya.

Table 2.3: The 1998 inflow drainage water chemistry means and standard deviations for Hogarth pit-lake. Concentrations are in mg·l^{-1} except alkalinity (mgCaCO$_3$·l^{-1}), conductivity (uS·cm^{-1}), hardness (calc.), and pH.

	Southeast		East		Seep	
	mean	std.dev.	mean	std.dev.	mean	std.dev.
Alkalinity	49.70	41.40	11.24	13.36	2.00	0.00
Conductivity	3250.00	558.30	2936.00	135.20	5092.00	1406.79
DIC	17.50	6.55	8.56	0.23	5.90	1.27
DOC	3.74	2.86	1.60	0.74	10.06	9.09
Hardness	2474.69	630.20	2234.96	188.80	2802.01	550.20
Cl^-	5.95	3.01	7.28	0.57	9.94	5.17
NO_3^-	0.35	0.04	0.08	0.03	0.28	.
SO_4^-	2704.49	396.22	2654.03	371.73	5601.74	2755.41
Ca	386.71	35.92	496.99	49.55	489.64	30.73
K	3.37	0.86	4.78	0.51	1.52	0.42
Mg	366.46	131.52	241.37	19.37	383.53	125.28
Na	14.33	6.66	11.14	0.68	6.16	0.95
Al	15.81	14.17	13.23	8.67	187.59	104.33
B	0.08	0.03	0.16	0.06	0.11	0.06
Co	0.11	0.15	0.11	0.02	1.06	0.58
Cr	0.014	0.008	0.028	0.015	0.931	0.608
Cu	0.08	0.07	0.06	0.03	1.45	0.86
Fe	1.55	2.17	14.99	5.53	168.48	142.80
Mn	8.40	9.49	16.79	2.50	25.96	12.20
Ni	0.65	0.70	0.61	0.15	3.70	2.11
Sr	1.22	0.19	0.82	0.06	0.74	0.16
Zn	0.09	0.08	0.10	0.03	0.97	0.53
pH	6.21	1.60	4.45	2.62	2.79	0.29
SiO_2	7.86	2.48	24.78	14.13	57.82	19.67
TKN	0.27	0.10	0.07	0.10	0.23	0.18
TDS	3675.33	1258.52	3383.84	341.21	6803.56	2404.98
TSS	28.80	3.32	85.16	11.28	28.52	10.22

Table 2.4. The 1999 inflow drainage water chemistry means and standard deviations for Hogarth pit-lake. Concentrations are in mg·l^{-1} except alkalinity (mgCaCO$_3$·l^{-1}), conductivity (uS·cm^{-1}), hardness (calc.), and pH.

	Southeast		Northeast		East		Seep	
	mean	std.dev.	mean	std.dev.	mean	std.dev.	mean	std.dev.
Alkalinity	153.18	86.94	201.58	10.30	2.78	1.82	2.02	0.04
Conductivity	3638.00	258.11	1711.80	47.30	2179.83	280.67	3694.00	378.66
DIC	24.91	13.32	28.97	18.33	0.54	0.62	3.00	1.20
DOC	3.73	1.39	2.62	1.02	2.69	1.93	6.72	2.44
Hardness	3155.46	291.14	1270.42	29.28	1493.51	170.00	2122.85	182.35
Cl^-	6.00	2.08	3.54	1.09	4.53	1.62	4.00	1.65
NO_3^-	0.44	0.16	0.12	0.01	0.10		< 0.01	
SO_4^-	2910.72	527.30	905.26	138.95	1475.75	376.38	3386.20	274.95
Ca	462.52	54.30	320.70	10.05	329.51	42.92	451.09	41.04
K	5.52	1.25	3.12	0.56	3.92	0.28	1.62	0.52
Mg	485.80	40.67	114.04	2.43	162.88	17.98	241.98	28.20
Na	23.08	3.17	4.28	0.22	8.69	0.51	5.24	0.55
Al	4.33	5.84	0.05	0.00	13.49	6.12	101.63	30.59
B	0.11	0.04	0.07	0.03	0.23	0.07	0.13	0.04
Co	0.03	0.05	0.01	0.00	0.08	0.02	0.49	0.13
Cr	0.012	0.01	0.005	0.00	0.092	0.097	0.626	0.178
Cu	0.04	0.04	0.01	0.00	0.06	0.03	0.83	0.27
Fe	0.33	0.40	0.03	0.01	32.16	28.41	118.03	46.02
Mn	3.36	5.49	0.00	0.00	10.59	2.51	11.21	2.35
Ni	0.26	0.37	0.02	0.00	0.51	0.15	2.00	0.50
Sr	1.25	0.69	0.53	0.02	0.64	0.07	0.59	0.10
Zn	0.03	0.03	0.02	0.02	0.09	0.03	0.47	0.12
pH	7.61	0.65	7.36	0.13	4.24	2.06	2.63	0.37
SiO_2	7.27	3.51	10.58	0.92	20.80	6.62	23.93	10.27
TKN	0.09	0.04	0.05	0.04	0.01	0.01	0.01	0.01
TDS	4058.04	924.43	1677.16	97.58	2362.60	294.15	4758.00	641.91
TSS	42.95	31.54	6.93	0.54	94.18	12.01	32.09	7.23

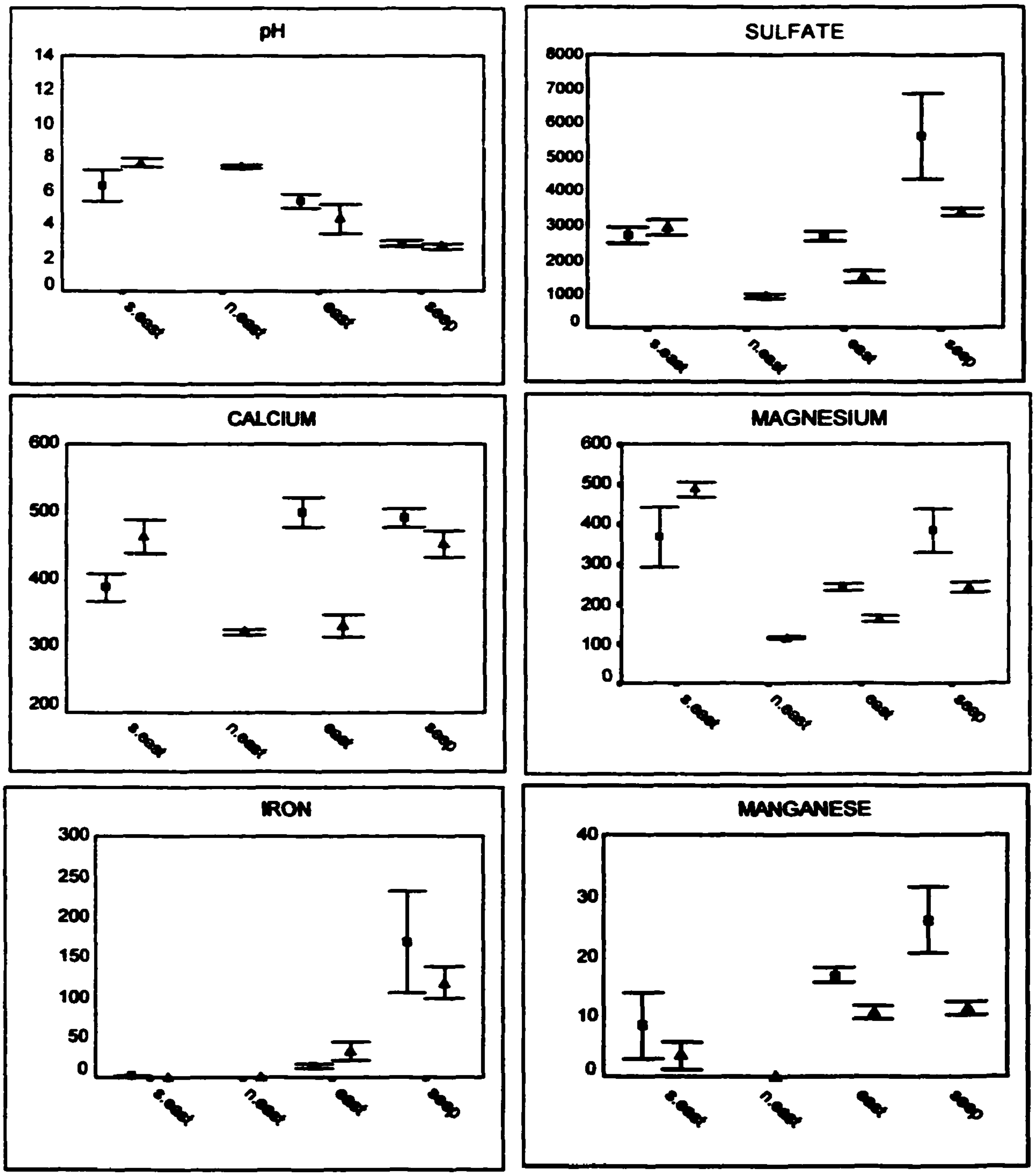

Figure 2.4: Inflow drainage water chemistry means and standard error for Hogarth pit-lake, comparing between 1998 (■) and 1999 (▲). Each of the parameters was measured in mg·l^{-1} with the exception of pH.

In 1998, the drainage was slowed or completely arrested due to two years (1997 and 1998) of low precipitation. Table 1.6 represents climate data modified from Environment Canada's Annual Summaries. Monthly temperatures were warmer in 1998. The drier climate, the slightly warmer temperatures, and their combined refraction from the pit rock walls and tailings areas magnified the aridity in the local environment. In 1998 warmer temperatures heated the retarded drainage contributing to increased leaching of SO_4, Cl, Ca, Mg, Na, Al, Co, Cu, Ni, Mn, Sr, and Zn at two of the three measured drainages (Table 2.3, 2.4, Figure 2.4). In 1999, there was higher rainfall (drainages flowed throughout the ice-free period), and temperatures were on average cooler compared to 1998.

2.3.4 Toxicity

There was no obvious sign of life in the pit's water column. Plankton tows conducted throughout the open water period were always empty. The LC_{50} conducted on *Daphnia magna* using full strength concentration resulted in 100% mortality within a 48hr period. A suspected toxin was arsenic since Ripley et al., (1996) stated that it is commonly found in iron ores. However, arsenic concentrations were below MDL, <0.002.

2.4 Discussion

2.4.1 Hogarth pit-Lake Water Chemistry

Water quality in Hogarth pit-lake was greatly affected by the scoured rock walls, proximal waste rock, and tailings piles which caused elevated concentrations of SO_4 (avg.1626.71 $mg \cdot l^{-1}$), Ni (avg. 0.065 $mg \cdot l^{-1}$), Ca (avg. 333.38 $mg \cdot l^{-1}$), Mg (avg. 207.61 $mg \cdot l^{-1}$), Mn (0.18 $mg \cdot l^{-1}$), TDS (avg. 2419.23 $mg \cdot l^{-1}$), conductivity (avg. 2286.21 $uS \cdot cm^{-1}$), and hardness (avg. 1681 calc.) (Table 2.1, 2.2). The likely source of sulfate was from lenses of pyrite found in a sedimentary formation along the hangingwall between the ore and the ashrock. This rock unit only outcropped sporadically throughout the Steep Rock Iron Mine Pits (Hogarth, Roberts, and Errington) (Shklanka, 1972). Nickel is commonly associated with such iron bearing rocks (Jaagumagi, 1992). Calcium was leached from the footwall carbonate which had beds up to several feet thick that varied in composition from calcite ($CaCO_3$) to dolomite ($CaCO_3 \cdot MgCO_3$) (Steep Rock Resources Inc., 1978). Mg was also derived from the dolomite as well as from the ashrock which is rich in olivine ($(Mg,Fe)_2SiO_4$) (Steep Rock Resources Inc., 1978). Manganese leached from the paint rock, which contains 3.78% Mn (Brawner Engineering Ltd. 1991). Similar pit-lake water quality was found in the Robinson District porphyry copper mine (Liberty Pit and Kimbley Pit) and the Getchell gold mine (South Pit, Center Pit, and North Pit) in Nevada. The Kimbley and Liberty Pits had calcareous rocks making up part of the wall rocks, but in the Liberty Pit acid leaching resulted in an acidic pit-lake with a pH of 3.2 (Miller et al. 1996). The Kimbley Pit was alkaline (pH 7.6), as was the North Pit in the Getchell Mine (pH 7.67) (Miller et al. 1996). Each of these aforementioned pits as well as the Berkeley Pit in Montana showed that pit-lakes in high sulfide rock tend to have poor quality water with elevated SO_4, Ca, Mg, Mn, and TDS. These pit-lakes may or may not be acidic dependent on the amount of limestone available to neutralize acid generated (Miller et al., 1996).

Although Hogarth pit-lake has a small surface area:depth which normally limits mixing (Hamblin et al., 1997), it did not appear to be meromictic in 1998. Instead most parameters appeared homogeneous with depth. Hogarth pit-lake had no oxygen minima which reflected its limited organic input due to the absence of aquatic life. Several morphologically similar pit-lakes have a well defined monimolimnion such as Berkeley Pit, Montana (Davis and Ashenberg, 1989); Brenda Pit, British Columbia (Hamblin et al. 1997); Caland Pit, Ontario (McNaughton et al. 1999); and the Gunnar Pit, Saskatchewan (Doyle and Runnells, 1997). In 1999, however, with increased rainfall, a meromix was evident in Hogarth pit-lake (Figure 2.3, Appendix D). Similarly, Mono Lake was monomictic in the early 1960's (Mason, 1967), and the late 1970's through early 1980's (Melack, 1983). In 1982-3, Mono Lake became meromictic through diversion of freshwater inflow. Meromixis persisted until 1988 and was characterized by a marked decrease in vertical mixing, (Jellison and Melack, 1993).

2.4.2 Variations between years, depth, and season

Several elements significantly differed in concentration between 1998 and 1999. These included alkalinity, conductivity, TDS, Ca, DIC, DOC, Cl, SiO_2, NO_3, Mn, TKN, and SO_4

(Table 2.1, 2.2). With the exception of the latter three parameters, the shallower waters in 1999 became less concentrated (Figure 2.3, Appendix D) due to increased rainfall during the second year of sampling. The "fresher" water eventually mixed by wind and wave action with the more dense shallow pit-lake water. The mixing would not extend too far in the water column as models by Hamblin et al., (1997) show that there is insufficient energy for a protected pit-lake to mix below 19 m. This dilution of the upper waters caused the significant variation with depth for most parameters.

Unlike the previous parameters, concentration of Mn (significant for both year and with depth) increased in surface waters due to increased leaching of the paint rock. Higher rainfall in 1999 either reacted directly on the soft rock material or perhaps the rise in surface water level came into contact with a new paint rock source. TKN concentrations increased at each depth in 1999 versus 1998. This increase is likely due to nitrogen in rainwater diffusing through the water column. In contrast, SO_4 concentrations decreased for each depth in 1999 also possibly due to the increased rainfall.

Thermal stratification caused variation in concentration with depth for water temperature, pH, and D.O. (Table 2.1, 2.2, Figure 2.2). Increased rainfall may have caused the D.O. concentration to be less constant in 1999 versus 1998 (Figure 2.2). Less dense waters have the ability to contain more dissolved oxygen. Mg and Na also were not significantly different between the two years but were with depth, as again their surface waters were more diluted in 1999 (Appendix D).

Every parameter (except Fe, Ni, Sr, TSS, and hardness) significantly varied in concentration with season (month sampled) (Table 2.1, 2.2, Figure 2.3, Appendix D) primarily due to ice-cover which prevented the dilution effect of precipitation, inhibited wind mixing, and decreased solar radiation. During winter the water column appeared to be vertically homogeneous (Figure 2.3, Appendix D). This may have been due to fall destratification and/or by the process of freezing in saline waters. Pure water freezes at a higher temperature causing the remaining water to increase in dissolved solids. In the latter process the pit-lake salinity (individual constituent chemistry) would increase. In 1999 this was especially noticeable in the shallower waters (Figure 2.3, Appendix D).

2.4.3 Influence of drainage and climate

During the ice-free period, both drainage and climate influenced the pit-lake water chemistry.

2.4.3.1 Drainage

At Hogarth, the seep and the east drainage exhibited AMD with pH between 2 and 4, and dissolved metal concentrations >1.0 $mg \cdot l^{-1}$ for Fe and Mn. Each of the drainages exhibited dissolved salt concentrations >100.0 $mg \cdot l^{-1}$ for SO_4, Ca, and Mg (Table 2.3, 2.4, Figure 2.4). Average concentrations for Al, Co, Cu, and Ni were only elevated at the seep (Table 2.3, 2.4).

Drainage concentrations between the two years of study varied inconsistently (Table 2.3, 2.4, Figure 2.4). Warmer and drier conditions in 1998 (Table 1.6) caused inflow to be retarded or arrested versus 1999. The east and seep drainages generally had higher levels in 1998 versus 1999 due to the stagnant flow. However, the southeast drainage had higher levels in 1999 possibly due to increased inflow contacting a broader surface area. Since the drainage water was denser than the pit-lake surface waters, the drainage inflow would sink below the surface water to a depth with an equivalent density layer. However, in all likelihood, these drainage inputs did not significantly influence the pit-lake water chemistry since the quantity of overland inflow was minimal in comparison to the capacity of the pit-lake. It is more likely that climate or ground water inflow had a more significant role.

2.4.3.2 Climate

The homogeneous water column in 1998 (Figure 2.3, Appendix D) could be related to climate. Low precipitation and slightly warmer summer temperatures in 1997 and 1998 (Table 1.6) increased the pit-lake water chemical concentrations through evapo-concentration. A similar situation was found by Miller et al. (1996) who determined that the concentration of all constituents in a pit lake in a semi-arid climate increased over time because of evapo-concentration. This also concurs with Williams (1996) who found that with a slight increase in aridity, less water enters a lake, causing the water level to fall and its salinity to rise.

The opposite effect occurred in 1999. The water in the shallower portion of Hogarth pit-lake became less concentrated and/or diluted with the comparably higher rainfall (Table 1.6). In a similar fashion, the Great Salt Lake decreased in salinity by more than 300 $g \cdot l^{-1}$ following a rise in freshwater level from 1960-89 (Williams, 1996).

2.4.4 Salinity tolerance for aquaculture and management

Hogarth pit-lake may be able to sustain aquaculture depending on what species of fish is cultivated. Although, the toxicity test was lethal for *Daphnia magna* this could have been due to osmotic stress. There are several aquatic fish species such as tilapia (*Oreochromis nilotica*), eels (*Anguilla rostrata*), and channel catfish (*Ictalurus punctatus*),that can tolerate Na and Cl dominated salinities greater than 5 ppt (Ackefors et al., 1994). Salinity tolerance for some species is a function of life cycle stage. For example, giant freshwater prawns (*Macrobrachium rosenbergii*) and mullets (*Catostomus* spp.) are catadromous; striped bass (*Morone saxatilis*), salmon, brook trout (*Salvelinus fontinalis*), and Arctic char (*Salvelinus alpinus*) are anadromous; and rainbow trout can be cultivated in full-strength sea water but they become intolerant upon reaching maturity (Ackefors et al., 1994). Sulfate dominated waters can also support aquaculture but ionic tolerance as well as osmotic affects need to be considered (Koel and Peterka, 1995).

2.4.4.1 Sulfate salinity tolerance for aquaculture

Hogarth pit-lake has an average SO_4 of 1626.71 mg·l^{-1}l, Ca of 333.38 mg·l^{-1}, Mg of 207.61 mg·l^{-1}, and TDS of 2419.23 mg·l^{-1}. Comparable SO_4 and TDS concentrations can be found in saline lakes of the Upper Great Plains in the United States and in Canada. In these lakes, growth of fishes is often excellent yet reproduction is restricted because salinity tolerance of embryos and sac fry is lower than that of adults (Atton, 1986). In Devils Lake, North Dakota, laboratory-based bioassays were conducted to determine concentrations of sodium sulfate salinities that limit the hatching success of several fish species. Survival to hatching (SH) was significantly lower ($p<0.05$) in sodium sulfate waters of >2400 mg·l^{-1} TDS then in freshwater of 200 mg·l^{-1} (Koel and Peterka, 1995). In waters of 200, 1150, 2400, 4250, and 6350 mg·l^{-1} TDS, walleye (*Stizostedion vitreum*) SH was 41, 38, 7, 1, and 0%; northern pike (*Esox lucius*) was 92, 68, 33, 2, and 0%; yellow perch (*Perca flavescens*) SH was 88, 70, 73, 0, and 0%; white sucker (*Catostomus commersoni*) SH was 87, 95, 66, 0, and 0%; common carp (*Cyprinus carpio*) SH was 71, 69, 49, 63, and 25% (Koel and Peterka, 1995).

However, even if adult fish species could tolerate the ion concentrations in Hogarth pit-lake there remain several factors that could impede their success. The iron floc and total dissolved solids present in the water column would irritate and/or clog the gills causing stress, which could lead to disease (OMOE, 1984). High levels of suspended solids would greatly reduce light penetration, photosynthesis, and production of natural food items (Ackefors, 1994). Elevated concentrations of either Ni (0.065 mg·l^{-1}) and/or Mg (207.61

$mg \cdot l^{-1}$) could be detrimental. Nickel, as per the Ontario Provincial Water Quality Objectives (PWQO), should not exceed 0.025 $mg \cdot l^{-1}$ (OMOE, 1994) and can be bioaccumulated by some organisms (CCREM, 1987). Concentrations of Mg greater than 125 $mg \cdot l^{-1}$ can have a cathartic and diuretic effect (Greenberg et al. 1992), and magnesium salts, especially $MgSO_4$, produce anesthesia in both invertebrates and vertebrates (Cole, 1994). Also, the "simple" fluctuation in salinity (expressed as TDS) between years may be too extreme for cultured aquatic species to acclimate.

2.4.4.2 Management

Hogarth pit-lake will eventually fill and outflow into the Seine River. The water quality and direction of this outflow are of major concern. The outflow would introduce contaminated water in excess of water quality guidelines into the surrounding watershed. Also, if the outflow is allowed to flow through the West Arm of Steep Rock Lake it may resuspend some of the deposited colloidal silty material.

Toxicity is also an issue. This study did produce some possible explanations for the lack of aquatic biota in Hogarth pit-lake such as the fluctuation in sulfate salinity, iron floc, and Ni and Mg levels, but the exact cause has yet to be identified. Fortunately for now, the calcium-magnesium hardness complexes with any heavy metals present and thereby reduces their toxicity (OMOE, 1984). If hardness decreases, toxic metal concentrations may occur. Determining the potential for such toxicity is paramount to future water management considerations of the Hogarth pit-lake.

There are two basic strategies for the overall management of Hogarth pit-lake: i) a "do nothing but monitor" approach which assumes that over time weathering reactions including AMD may subside and pit-lake ions precipitate; or ii) a pro-active approach. Pro-active options may be: i) deposit the proximal tailing piles into the pit-lake to reduce ARD. Feasby (1993) suggested that water coverage of reactive tailings and exposed rock surfaces rendered some minerals unreactive. Underwater tailings disposal promotes the removal of suspended solids from slurry through settling, minimizes the dissolution of metals from sediments, and renders sulfide minerals unreactive by limiting the access of oxygen to reactive solid components. The use of submerged impoundments for long term storage of potentially acid generating tailings is commonly practiced in the Canadian gold mining industry (Scott and Ingles, 1987); ii) continue to revegetate the exposed areas including waste piles preferably with native plants. Revegetation would reduce erosion, reduce direct sun exposure thereby alleviating influence of temperature on weathering reactions, and plants such as *Typha* would absorb and trap high concentrations of elements; iii) dump large quantities of organic matter into the pit-lake. Under oxidizing conditions at neutral pH, concentrations of positively charged trace metal species are controlled by adsorption on substrates such as clays and organic matter (Miller et al., 1996). This solution may cause anoxic conditions and may alter the pH but should alleviate the detrimental ion content; and iv) pump in freshwater from nearby sources. Quickly filling the pit-lake would dilute the high ion concentrations or trap the saline waters at depth.

General Conclusions

Although Caland and Hogarth pit-lakes are located only one kilometer apart, their water quality contrasted considerably. Caland pit-lake appears to be developing into meromictic lake with an anoxic monimolimnion. Hogarth pit-lake, on the other hand, has extremely elevated dissolved solids promoting its sulfate-saline condition and has no aquatic fauna despite the entire water column being well oxygenated.

The surrounding surficial geology and climate greatly influenced the water quality in Hogarth pit-lake. The pyritic lenses in the rock unit surrounding Hogarth pit-lake increased sulfate levels and consequently TDS levels in the water column. Evapo-concentration in 1998 promoted a chemically homogenous water column, and increased rainfall in 1999 induced meromictic conditions. In comparison, the impact of the recently scoured rock faces on Caland's pit-lake water chemistry was less noticeable, and surface water inflow in 1998 exceeded evaporation.

Drainage had considerable influence on Caland pit-lake. Inflow of freshwater was much greater into Caland pit-lake and resulted in meromictic conditions, with an anoxic monimolimnion, and water quality in the mixolimnion within the Provincial Water Quality Objectives. Between 1998 and 1999 surface water level increased 5 m. Although drainage into Hogarth pit-lake was more variable and degraded, the quantity was minimal.

Statistically significant variations occurred between the two pit-lakes for metal, anion, cation, conductivity, hardness, and dissolved oxygen concentrations. Variations in precipitation and evaporation between the two sampling years resulted in increased or decreased ion concentrations for the same seasons or depths within the two pit-lakes. Also, in Caland pit-lake, the expanding fish farm influenced the water quality.

The fish farm expansion in Caland pit-lake resulted in an increase in phosphorus and ammonium levels, and in dissolved oxygen consumption. Prior to the recent expansion, the organic matter inputs from the farm and the pit-lake morphology promoted the meromix and resulted in only 20m of dissolved oxygen (greater than 4.0 mg/l) remaining. Further increases in fish production in this pit-lake should be reassessed. In Hogarth pit-lake, aquaculture would have to be limited to saline tolerant species.

Both Hogarth and Caland pit-lakes as well as the surrounding environs require monitoring. As yet, the exact cause of toxicity in Hogarth pit-lake has not been determined, and the water quality in Caland pit-lake is declining. Also, in Caland pit-lake, with increasing surface area there is potential for wind activity to influence the unstable chemocline and effect a turnover. Furthermore, with the increasing surface area the mixolimnion layer may decrease. As suggested by Miller et al., (1996), because of the immensity of pit-lakes, any remediation of water quality may be unrealistically expensive. The water quality in pit lakes are effectively permanent and may affect utilization of water and land resources for the foreseeable future. Banks et al., (1997) suggests that mine-water problems may be

addressed by isolating the contaminant source, by suppressing the reactions releasing contaminants, or by active or passive water treatment. Treatment such as galvanic suppression, applications of bacteriacides, neutralising or reducing agents (pulverised fly ash-based grouts, cattle manure, whey, brewers' yeast) require additional research.

References

Ackefors, H., and M. Enell. 1990. Discharge of nutrients from Swedish fish farming to adjacent sea areas. Ambio. 19:28-35.

Ackefors, H., J.V. Huner, and M. Konikoff. 1994. Introduction to the General Principles of Aquaculture. Haworth.

Atton, F.M. 1986. Fish in Saskatchewan endorheic systems and saline waters. *In* Evaluating saline waters in a plains environment. *Edited by* D.T. Waite. Canadian Plains Research Center, University of Regina, Regina, Sask. Pp. 65-74.

Axler, R., C. Larsen, C. Tikkanen, M. McDonald, and G. Host. 1992. Limnological Assessment of Mine Pit Lakes for Aquaculture Use. Technical Report: NRRI/TR-92/03. Center for Water and the Environment, Natural Resources Research Institute, University of Minnesota, Duluth.

Axler, R., S. Yokom, C. Tikkanen, J. Henneck, and M. McDonald.1995. Effects of aquaculture on mine pit lakes near Chisholm, MN: restoration of Twin City-South pit lake by fallowing and status of Fraser pit lake. Technical Report: NRRI/TR-95-04. 21+pp.

Axler, R., C. Larsen, C. Tikkanen, M. McDonald, S. Yokom, and P. Aas. 1996. Water quality issues associated with aquaculture: A case study in mine pit lakes. Water Environ. Res. 68:995-1011.

Banks, D., P.L. Younger, R.T. Arnesen, E.R. Iversen, and S.B. Banks. 1997. Mine-water chemistry: the good, the bad and the ugly. Environ. Geol. 32(3):157-174.

Bergheim, A., J.P. Aabel, and E.A. Seymour. 1991. Past and present approaches to aquaculture waste management in Norwegian net pen culture operations. in C.B. Cowey and C.Y. Cho (eds.) Nutritional Strategies in Management of Aquaculture Waste. Proc. First International symposium on Nutritional Strategies in Management of Aquaculture Waste. 5-8 June 1990, Guelph, Ontario, CAN. pp117-136.

Boyd, C.E., and W.W. Walley. 1972. Studies of the biogeochemistry of boron. I. Concentrations in surface waters, rainfall and aquatic plants. The American Midland Naturalist. 88(1):1-14.

Brawner Engineering LTD. 1991. Report to Ministry of Natural Resources Thunder Bay,Ontario on Hazard Evaluation – Steep Rock and Caland Mine Areas. 69pp.

Brierley, C.L. 1978. Bacterial leaching. CRC Critical Rev. Microbiol. 6:207-262.

Bureau, D.P., and C.Y. Cho. 1999. Nutritional strategies for the management of aquaculture wastes. *In* 1999 Great Lakes Fishery Commission Roundtable. Addressing Concerns for Water Quality Impacts from Large-Scale Great Lakes Aquaculture ISBN 1-894280-10-5, Windsor, Ontario pp.28-31.

CCREM (Canadian Council of Resource and Environment Ministers), 1987. Canadian Water Quality Guidelines. Water Quality Branch, Inland Waters Directorate, Environment Canada, Ottawa, Ont., Canada.

Chapman, G. 1997. Assessment of the Biological Capacity of Caland Lake. Unpublished Manuscript Prepared for Snow Lake Fish Farms. 16pp.

Cloud, P. 1973. Paleoecological significance of the banded iron-formation. Econ. Geol. 68:1135-1143.

Cole, G.A. 1994. Textbook of Limnology 4th ed. Waveland Press Inc: Illinois. 412pp.

Cornel, G.E., and F.G. Whoriskey. 1993. The effects of rainbow trout (*Oncorhynchus mykiss*) cage culture on the water quality, zooplankton, benthos and sediments of Lac du Passage, Quebec. Aquaculture 109:101-117.

Davis, A. and D. Ashenburg. 1989. The aqueous geochemistry of the Berkeley Pit, Butte, Montana, U.S.A. Appl. Geochem. 4:23-36.

Doyle, G.A., and D.D. Runnells. 1997. Physical limnology of existing mine pit lakes. Mining Engineering 49:76-80.

Environment Canada. 2000. Environment Canada's Atikokan-Mine Centre Weather Station for 1997, 1998, and 1999.

Environmental Protection Agency. 1988. Standard operating procedure for the analysis of sulfide in water method 376.1 NS (Tirtrimetric, Iodine). Region 5 Central Regional Laboratory. Illinois, USA. 14pp.

Environmental Protection Services. 1996. Environmental Test Method: Reference Method for Determining Acute Lethality of Effluents to *Daphnia magna*. Minister of Supply and Services Canada. Beauregard Printers Ltd.

Eugster, H.P., and L.A. Hardie. 1978. Saline lakes. *In* A. Lerman, Ed. Lakes: Chemistry, Geology, Physics. Springer-Verlag, New York. pp.237-293.

Feasby, D.G. 1993. The mine environment neutral drainage (MEND) program. CIM Bull.86(971):71.

Feth, J.H. 1971. Mechanisms controlling world water chemistry: Evaporation-crystallization process. Sci. 172:870.

Gale, P. 1999. Water quality impacts from aquaculture cage operations in the LaCloche, North Channel of Lake Huron. *In* 1999 Great Lakes Fishery Commission and Great Lakes Water Quality Board Roundtable. Addressing Concerns for Water Quality Impacts from Large-Scale Great Lakes Aquaculture ISBN 1-894280-10-5, Windsor, Ontario pp.51-57.

Gibbs, R.J. 1970. Mechanisms controlling world water chemistry. Sci. 170:1088-1090

Gould, W.D., G. Bechard, and L. Lortie. 1994. The nature and role of microorganisms in the tailings environment. *In* Short Course Handbook on Environmental Geochemistry of Sulfide Mine-Wastes. ed. J.L. Jambor and D.W. Blowes. Mineralogical Association of Canada. p185-199.

Greenberg, A.E., L.S. Clesceri, and A.D. Eaton. 1992. Standard Methods for the Examination of Water and Wastewater, 18th edition EPS Group Inc, Hanover Maryland.

Habitat Advisory Board of the Great Lakes Fishery Commission and the Great Lakes Water Quality Board of the International Joint Commission. 1999. Addressing Concerns for Water Quality Impacts from Large-Scale Great Lakes Aquaculture ISBN 1-894280-10-5, Windsor, Ontario 78pp.

Hamblin, P.F., C.L. Stevens, and G.A. Lawrence. 1997. Vertical transport in Brenda Mines pit lake, p. 367-384. In Proc. 4th Intl. Conf. On Acid Rock Drainage, Vancouver, B.C. V.1.

Jaagumagi, R. 1992. Development of the Ontario Provincial Sediment Quality Guidelines for Arsenic, Cadmium, Chromium, Copper, Iron, Lead, Manganese, Mercury, Nickel, and Zinc. Ontario Ministry of Environment Water Resources Branch. ISBN 0-7729-9249-5.

Jellison, R. and J.M. Melack. 1993. Meromixis in hypersaline Mono Lake, California. 1.Stratification and vertical mixing during the onset, persistence, and breakdown of meromixis. Limnol. Oceanogr. 38:1008-1019.

Jorgensen, B.B., J.G. Kuenen, and Y. Cohen. 1979. Microbial transformations of sulfur compounds in a stratified lake (Solar Lake, Sinai). Limnol. Oceanogr. 24:799-822.

Kilham, P. 1975. Mechanisms controlling world water chemistry based on data for African lakes and rivers. Proc. Int. Symp. Geochemistry of Natural Waters. Burlington, Ontario. 5pp.

Koel, T.M., and J.J. Peterka. 1995. Survival to hatching of fishes in sulfate-saline waters, Devils Lake, North Dakota. Can. J. Fish. Aquat. Sci. 52:464-469.

Kuyucak, N. 1999. Acid mine drainage prevention and control options. *In* International Mine Water Association Congress on Mine, Water & Environment, Sevilla, Spain Vol.II p599-606.

Lakehead University Environmental Lab. 1998. Standard Operating Procedures.

Lakehead University Environmental Lab. 1999. Standard Operating Procedures.

Lakehead University Environmental Lab. 2000. Standard Operating Procedures.

Lindsay, D. 1999. Owner of Snow Lake Fish Farm. Personal communications.

Lindsay, D. 2000. Owner of Snow Lake Fish Farm. Personal communications.

Lundgren, D.G. and M. Silver. 1980. Ore leaching by bacteria. Ann. Rev. Microbiol. 34:263-283.

Mason, D.T. 1967. Limnology of Mono Lake, California. Univ. Calif. Publ. Zool. 83, 110pp.

Melack, J.M. 1983. Large, deep salt lakes: a comparative limnological analysis. Hydrobiol. 105:223-230.

McNaughton, K., P.F. Lee, D. Lindsay, and M.P. Sudbury. 1999. The limnology of an open pit fish farm. *In* International Mine Water Association Congress on Mine, Water & Environment, Sevilla, Spain Vol.1 p169-175.

Miller, G.C., W.B. Lyons, and A. Davis. 1996. Understanding the water quality of pit lakes. Environ. Sci. Tech./News. 30(3):118A-123A.

Moccia, R.D., and D.J. Bevan. 1999. Aquastats 1998. Part 1: Ontario Aquaculture Production in 1998. University of Guelph. Report No. 99-002. 2pp.

Mortimer, C.H. 1941. The exchange of dissolved substances between mud and water in lakes. (parts I & II). J. Ecol. 29:280-329.

NCC. 1990. Fish farming and the Scottish freshwater environment. NCC Contract No. HF3-03-450. Nature Conservancy Council, Edinburgh, Scotland. 285p.

Ojakangas, R.W. and C.L. Matsch. 1982. Minnesota's Geology. University of Minn. Press. 255p.

Ontario Ministry of Environment. 1984. Water Management: Goals, Policies, Objectives and Implementation Procedures.

Ontario Ministry of Environment. 1988-1993. Unpublished data. Thunder Bay, Canada

Ontario Ministry of Environment. 1992. Development of the Ontario Provincial Water Quality Guidelines for arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury nickel, and zinc. Water Resources Branch. ISBN#0-7729-9249-5. 10pp.

Ontario Ministry of Environment and Energy, 1994. Water Management:Policies, Guidelines, Provincial Water Quality Objectives

Ontario Ministry of Northern Development and Mines. 1994. Rock On Series 1, 89p.

Pye, E.G. 1968. Geology and scenery, Rainy Lake and east to Lake Superior. Ont. Dept. Mines, Geol. Guide Book 1, 114pp.

Ripley, E.A., R.E. Redmann and A.A. Crowder. 1996. Environmental Effects of Mining. St. Lucie Press: Florida. 356pp.

Ritchie, A.I.E. 1994. The waste-rock environment. *In*: Jambor JL and Blowes DW (Eds),Short course handbook on environmental geochemistry of sulfide mine-wastes. Mineral Association Canada 22:133-162.

Rosenthal, H., D. Weston, R. Gowen, and E. Black. 1988. Report of the ad hoc StudyGroup on "Environmental Impact of Mariculture". ICES, Cooperative Research Report 154, Copenhagen, Denmark, 83p.

Scott, J.D. and J. Ingles.1987. State-of-the-Art Processes for the Treatment of Gold Mill Effluents. Mining, Mineral, and Metallurgical Processes Division, Industrial Programs Branch, Environment Canada, Ottawa, Ontario, March.

Shklanka, R. 1972. Geology of the Steep Rock Lake area district of Rainy River. Ont. Dept. Mines and Northern Affairs. Geol. Rep. 93. 114pp.

Stallard, R.F. 1980. Major element geochemistry of the Amazon River system. Ph.D. Dissertation, Massachusetts Institute of Technology.

Stallard, R.F., and J.M. Edmond. 1981. Geochemistry of the Amazon. I. Precipitation chemistry and the marine contribution to the dissolved load at the time of peak discharge. J. Geophys. Res. 86:9844-9858.

Stangenberg-Oporawska, K. 1967. Sodium compounds in carp-pond waters in Poland. Pol. Arch. Hydrobiol. 14:11-17.

Steep Rock Mines. 1943. The Steep Rock iron ore deposits. A story of patient investigation and development. Can. Mining. J. 63:437-444.

Steep Rock Iron Mines Ltd. 1978. Environmental Plan.

Steep Rock Resources Inc. 1986. Report on surrender of mining claims.

Stevens, C.L., and G.A. Lawrence. 1998. Stability and meromixis in a water-filled mine pit. Limnol. Oceanogr. 43(5):946-954.

Stumm, W., and J.J. Morgan. 1981. Aquatic Chemistry. An introduction emphasizing chemical equilibria in natural waters. 2nd Edition. J. Wiley & Sons, New York. 780pp

Sudbury, M. 1998. Personal communication. Falconbridge Environment.

Walker, K.F., and G.E. Likens. 1975. Meromixis and a reconsidered typology of lake circulation patterns. Verhandlungen Internationale Vereiniguang fur Limnologie. 19:442-458.

Wetzel, R.G. 1983. Limnology 2nd ed. Saunders College Publishing: Toronto, Ontario.767pp.

Wetzel, R.G., P.H. Rich, M.C. Miller, and H.L. Allen. 1972. Metabolism of dissolved and particulate detrital carbon in a temperate hard-water lake. Mem. Ist. Ital. Idrobiol. 29(suppl.):185-243.

Williams, W.D. 1978. Limnology of Victorian Salt Lakes, Australia. Verh. Int. Ver. Limnol. 20:1165-1174.

Williams, W.D. 1996. What future for saline lakes? Environ. 38(9):13-39.

APPENDIX

Appendix - A

THE LIMNOLOGY OF AN OPEN PIT FISH FARM

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THE LIMNOLOGY OF AN OPEN PIT FISH FARM

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ABSTRACT

The unique geology and physical features of an open pit mine site that contributes to the successful establishment of an aquaculture operation were examined. Located under Steep Rock Lake near Atikokan, Ontario, the high grade hematite ore was exposed by diverting water from the lake and excavating 205 million cubic meters of overburden. Open pit mining took place from 1944 to 1979 and over 85 million tonnes of ore were extracted from the Caland and Steeprock Pits, located in the east and middle arms respectively of the original lake bed. After closure, the pits were allowed to fill with water from rainfall, snowfall and groundwater seepage reaching present depths of over 150 meters. The occurrence of limestone deposits at the site counters the production of acid from waste rock and the water in the pits remained above neutrality. In 1988, Snow Lake Fish Farms began raising trout and salmon species in the Caland Pit. Detailed water quality testing was conducted during 1998. Water chemistry in the two pits was found to be largely dependent on the type of proximal waste rock and surficial geology. The water in the pits differed in metal, anion, cation, conductivity, hardness and dissolved oxygen concentrations. Although there was nutrient loading from excess fish food and feces, phosphorus levels were not elevated in the water column. It was hypothesized that the increases of phosphorus in the Caland Pit were suppressed by the deposition of phosphorus with iron compounds in this deep meromictic basin. The current expansion of the fish farm operation is not expected to increase phosphorus concentrations in the water column, but the volume of usable water may be diminished due to the increased oxygen demand.

INTRODUCTION

Newly scoured surfaces from open pit mining presents a unique opportunity to study the effects of the chemically active rock walls, waste rock, and tailings on these man-made closed lake systems. Generally the composition of the water depends on the minerals present in the exposed rock, their relative abundances, their weatherability, the actual weathering reactions involved, and the drainage condition (Eugster and Hardie, 1978). The utilization of abandoned open pit mines for aquaculture adds additional new chemical variables that increases lake productivity in these unsettled environments.

In northern Minnesota's Mesabi Range, intensive aquaculture operations were established in several abandoned open pits that had moderately hard water (Axler et al., 1996a). In Quebec, a rainbow trout rearing facility was located at Lac Du Passage and was also characterized by slightly basic water (Cornel and Whoriskey, 1993). The acid buffering capacity of these flooded pits can therefore be suitable for fish cultivation. Unfortunately a second problem that often arises from the actual aquaculture operation is the development of anoxic conditions in the hypolimnion caused by the BOD of fish food and feces that settles to the bottom of the pits. In the absence of oxygen, P is released at the sediment:water interface, mixed throughout the water column at spring and fall turnover and results in eutrophic conditions. Any discharge from the pits will adversely affect the water quality of the receiving water body. This was essentially the situation in Minnesota and the inability of the aquaculture company to meet discharge standards ultimately caused its bankruptcy (Axler et al., 1992).

Non compliance of environmental regulations has therefore limited the use of open pits for aquaculture. Nevertheless, the premise of utilizing an abandoned mine site for aquaculture remains attractive from both an economic and social sense. The challenge is to devise a set of criteria for identifying suitable open pits for this purpose and to develop the needed techniques that will limit any detrimental effects on the environment. Falconbridge Inc., Lakehead University and Snow Lake Fish Farms have embarked on a major research program that should result in the successful utilization of abandoned open pits for aquaculture.

GEOLOGY OF THE STUDY SITE

The Study Site is located at the southern margin of the granite-greenstone Wabigoon subprovince of the superior province of the Canadian Shield (Ontario Ministry of Northern Development and Mines, 1994). The Steep Rock area (Fig.1) 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 bodies. The ore is bordered by "Paint Rock", a soft clay-like material eroded from the limestone, and by "Ash Rock" formed from volcanic ash. 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 760m, width of 50m, and length of over 4km. The ore consisted mostly of geothite 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 action, 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 when 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 give credence to this hypothesis.

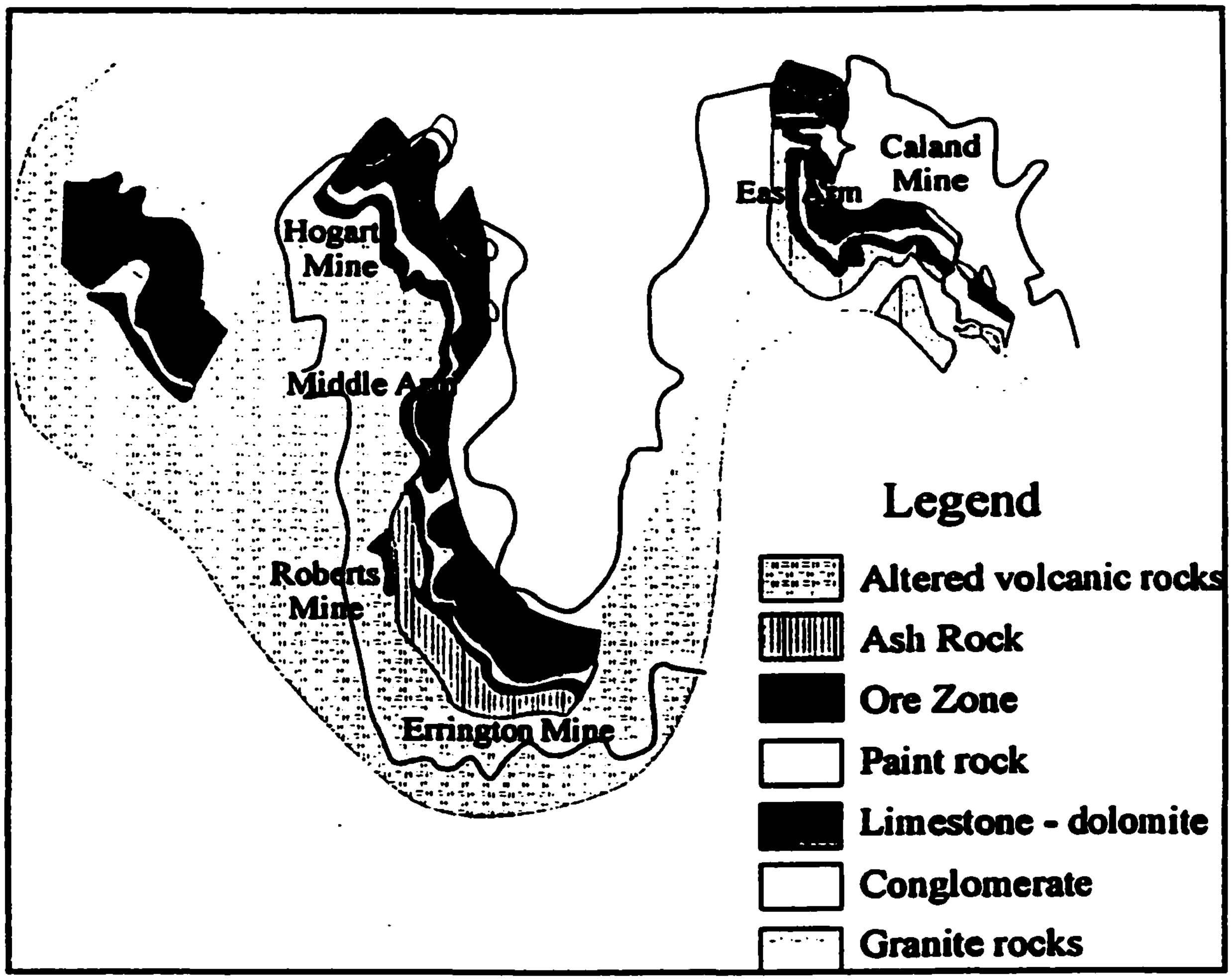


Figure 1. Geological map of the former Steep Rock Lake. The main ore bodies were displaced from each other along large faults (modified from Pye (1968)).

MINING HISTORY OF STUDY SITE

The occurrence of iron bearing rocks in the Atikokan region was first recognized by Smyth in 1891, but it was not until 1930 that the ore bodies were located below Steep Rock Lake (Steep Rock Mines, 1943). Since the iron ore body was located under Steep Rock Lake, the lake had to be drained. This was accomplished by diverting the inlet, the Seine River, to flow through Finlayson Lake and installing a series of dams (Fig. 2). The Middle and East Arms of the lake were pumped dry and about 205 million m^3 of overburden were removed. Steep Rock Iron Mines mined the Middle Arm of Steep Rock Lake (Hogarth, Errington, and Roberts Pits), while Caland Ore mined the East Arm of Steep Rock Lake (Caland Pit). Both companies had ore pelletizing plants. From 1944 until 1979 when the mines were closed, over 85 million tons of ore were removed (Pye, 1968). After mining ceased, the rights to the land reverted to the Government of Ontario and the pits began to naturally fill with water.

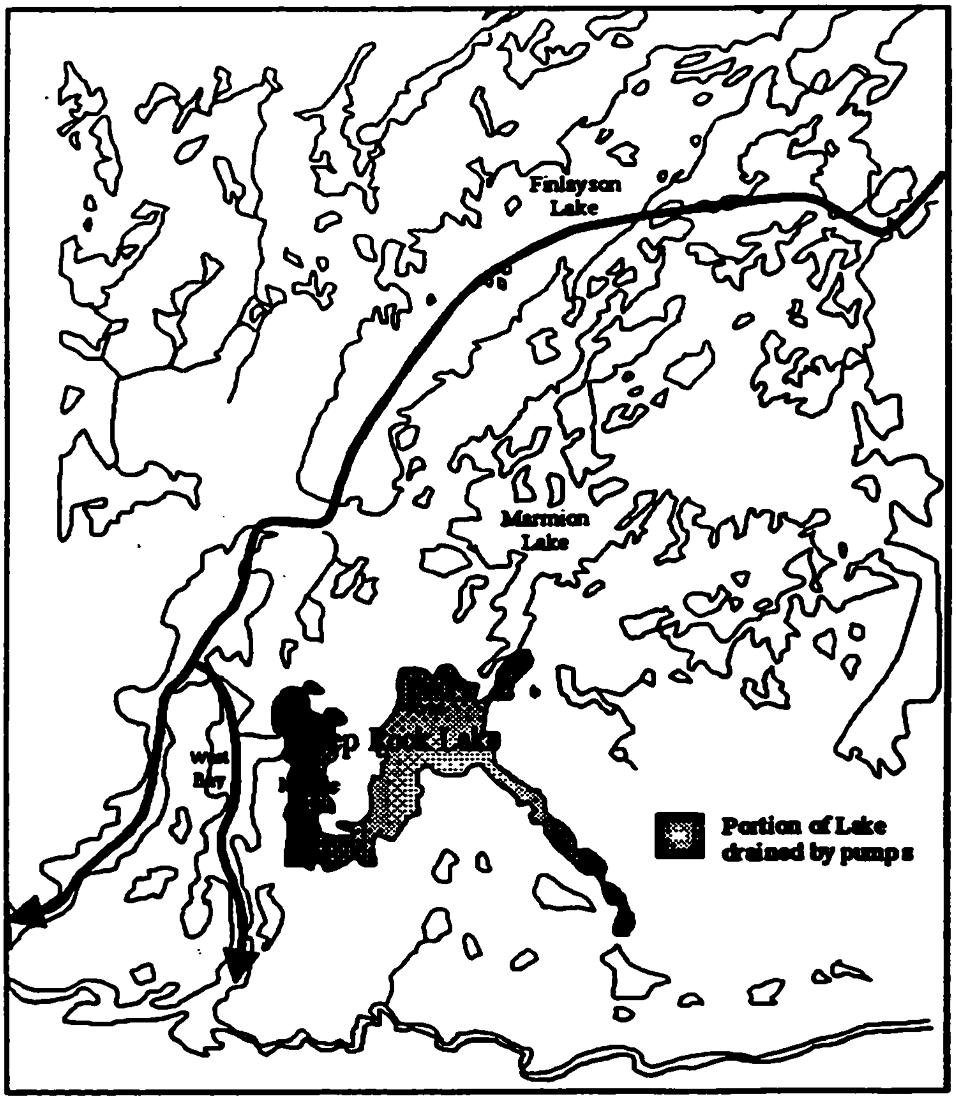


Figure 2. Location of open pits (Caland and Hogarth) that were utilized after Steep Rock Lake was drained. The arrows indicate the flow of water after Steep Rock Lake was drained. (modified from Pye (1968)).

SNOW LAKE FISH FARM

Snow Lake Fish Farm was established in 1988 in the Caland Pit. It uses cage culture to raise primarily Rainbow Trout. The cages are extended in the water column to a depth of 20m. The fish are raised throughout the year but growth occurs primarily in the summer. The normal procedure is to place trout fingerlings in smaller 7.6m diameter cages in the spring where they remain until they are 10cm in length. The fish are then transferred first to cages 15m in diameter and finally to 23m diameter cages where they are grown to market size (1-2kg). The larger cages have a capacity of 20,000kg in the summer and 40,000kg in winter. Processing of the fish was done on site in a floating building until recently when a processing plant was built in the nearby town of Atikokan. Until 1998, maximum production on the lake was 52,000kg/yr. In 1998, a major expansion of the facility was undertaken to gradually increase production to 225,000kg/yr (Snow lake Fish Farms, 1997). The major concern with the fish farm is the effect on water quality caused by wasted feed and fish excretion. An assessment of the carrying capacity of the pit by Chapman (1997) determined that increased P inputs could have a severe impact on water quality and lead to oxygen depletion.

STUDY SITE

The study site consists of an open pit used for aquaculture and a similar nearby open pit that has no aquaculture. These pits are located at the former site of Steep Rock Mines at Steep Rock Lake (48° 48'N, 91°39'W) near Atikokan, Ontario, Canada (Fig. 2). Caland Pit, located in the former East Arm of the lake, has an active aquaculture operation. It currently has a depth of 180m with an area of 1.5km^2 and acts as the test site. Hogarth Pit, located in the northern part of the former Middle Arm, contains no fish and acts as our control site. Hogarth has a depth of approximately 155m and an area of 1.6km^2. Both pits continue to increase in depth from rainfall and runoff, and, by the year 2030, Caland Pit will be connected to Hogarth Pit. At that time, the maximum depth of the newly connected pits will be 425m with a combined area of 13.4km^2(Chapman, 1997). Currently, the Caland and the Hogarth Pits have a bowl/cone shaped basin morphology with very low surface area to depth ratio. These lakes are steep sided and well sheltered from winds thus promoting a meromictic environment.

METHODOLOGY

The Ontario Ministry of Environment (MOE) collected water samples from both the Caland and Hogarth pits from 1988 to 1993 inclusive. Their water chemistry analysis followed the Ontario Ministry of Environment Laboratory Guidelines (MOE, 1988-93).

More currently, June and July 1998, Caland and Hogarth were sampled for the corresponding water chemistry parameters at depths: epilimnion (1m), metalimnion (approx. 6m), hypolimnion (30m) and (1m) off-bottom (175m and 155m respectively). The water samples were analyzed in the Lakehead University Environmental Laboratory (1998) following their Standard Operating Procedures including Quality Assurance/Quality Control protocols. The Caland upper water parameter measurements were quite similar and thus combined over the two month sampling period, similarly the lower two sampling depths were combined over the same period. In Hogarth, there was no significant difference among the depths thus the 4 sampling depths were combined over the two months. A combined mean summary of the earlier MOE and current data are included in Table 1.

RESULTS AND DISCUSSION

The MOE results (Table 1) show that Hogarth had higher concentrations of Ni, Fe, Mn, Conductivity (EC-uS.cm^{-1}), hardness, Mg, Ca, Na, K, and particularly SO_4 compared to Caland. In the summer of 1992, four years after fish farming commenced, dissolved O_2 levels were similar in both pits at a depth of 1m, but at the 10 and 40m depths, Caland had substantially lower oxygen levels (5.0mg/l) than Hogarth (9.0mg/l) (Fig. 3). Combined average P concentrations were below method detection limit, 5ug/l, but NO_3^- and TKN concentrations were substantially higher in the Caland pit. Recent findings are similar in the comparison between the two pits (Table 1).

Currently, the majority of the measured parameters in the upper waters of Caland have decreased in concentration over the past several years even with the increase in farming production. Alternately, the lower waters have increased in anion, cation, and metal concentrations, especially NO_3 and SO_4 (Table 1), but has severely decreased in dissolved oxygen (Fig. 3) compared to the MOE data. The characteristics of the main water quality concerns associated with the fish farming practice are discussed below:

Phosphorus

At the fish farm there does not appear to be a problem as yet with the phosphorus levels as they remain below method detection limits (5ug/l). Chlorophyll readings are also reflective of an oligotrophic environment. Essentially, under oxidized conditions, P precipitates with FeOH complexes, but under anoxic conditions the P is released as Fe^{+++} is converted to Fe^{++} (Mortimer, 1941). In the open pit situation, the continuous supply of oxidized iron from the waste rock should remove P from the water column, but this P is becoming available again for biological uptake and possible eutrophication risk as the redox boundary is currently at the 20m depth. The released iron rapidly forms an insoluble bond with sulfur at these low redox levels and precipitates (Wetzel, 1983). Axler et al., (1995) suggested that although P-release from anoxic sediments would probably increase it would not likely impact water quality since the increased P-levels would occur far below the euphotic zone (typically 10-20m).

Nitrogen

The free ammonium levels have remained quite low at the fish farm (0.02mg/l) despite intense feeding during the fish growth season. Bergheim et al., (1991) stated that only 25-30% of the nitrogen and phosphorus in feed is typically retained in the fish, the remainder being lost to the environment. High rates of nitrogen loading as excreted ammonium may contribute to excessive algal growth, increased rates of oxygen depletion (via nitrogenous BOD), and potentially to unionized ammonia toxicity (Axler et al., 1996). These low ammonium levels leads to a belief that nitrification must be occurring fairly rapidly in the Caland system as neither ammonium toxicity or algal growth are of immediate concern.

Total Kjeldahl Nitrogen and NO_3 levels (Table 1) have decreased in the upper waters of Caland compared to earlier findings which may be due to increased hypolimnion anoxia. Axler et al., (1995) suggested a management option of providing some control of NO_3 buildup by denitrification would require allowing bottom waters to become anoxic for a

Table 1. Mean values collected and pooled from MOE records in June & July 1988-93, compared to recent data collected from same months in 1998. All parameters are in mg/l except for pH and EC (electrical conductivity umhos/cm); (* less than detection limits).

	1	2	3	4	5	6
Alkalinity	125.45	141.39	169.55	158.94	95.74	65.91
Conductivity	724.25	863.09	1131.58	981.13	2438.08	2081.54
DIC	33.98	na	63.91	na	29.39	na
DOC	4.44	na	4.31	na	2.52	na
Hardness	360.6	490	616.71	552.38	1680.62	1382.08
Cl	8.45	12.43	13.73	13.06	18.04	15.49
NO_3	3.04	3.45	5.3	3.73	1.61	1.1
Ca	83.72	106.36	133.95	122.13	337.14	295
K	4.61	4.62	6.13	4.76	6.98	5.52
Mg	36.8	54.45	68.54	60.13	208.35	166.15
Na	11.91	16.27	18.62	18	24.99	23.08
Al	*	*	*	*	0.02	0.02
B	0.06	0.05	0.06	0.06	0.05	0.05
Be	*	*	*	*	*	*
Cd	*	*	*	*	*	*
Co	*	*	*	*	*	*
Cr	*	*	*	*	*	*
Cu	*	*	*	*	0.01	*
Fe	0.02	0.06	0.03	0.04	0.4	0.29
Mn	0.01	0.04	0.13	0.01	0.16	0.34
Mo	*	*	*	*	*	*
Ni	*	*	*	*	0.07	0.1
Pb	*	*	*	*	*	*
Sr	0.63	0.3	1.05	0.25	1.48	0.28
V	*	*	*	*	*	*
Zn	0.04	*	0.01	*	0.01	*
SO_4	250.92	327.36	475.92	365.25	1902.54	1314.92
NH_4	0.02	*	0.01	*	0.02	*
pH	7.98	8.2	7.08	7.73	7.12	7.39
SiO_2	2.16	3.32	9.41	3.98	9.58	3.53
TDS	435.7	724.45	835.6	834.88	2492.46	2138.38
TKN	0.08	0.58	0.1	0.36	0.02	0.15
TP	*	*	*	*	*	*
TSS	*	*	*	*	*	*

Caland Upper June&July, 1998	1
Caland Upper J/J 88,90,92,93	2
Caland Lower June&July, 1998	3
Caland Lower J/J 88.90.92,93	4
Hogarth June&July 1998	5
Hogarth June/July 88,90,92,93	6

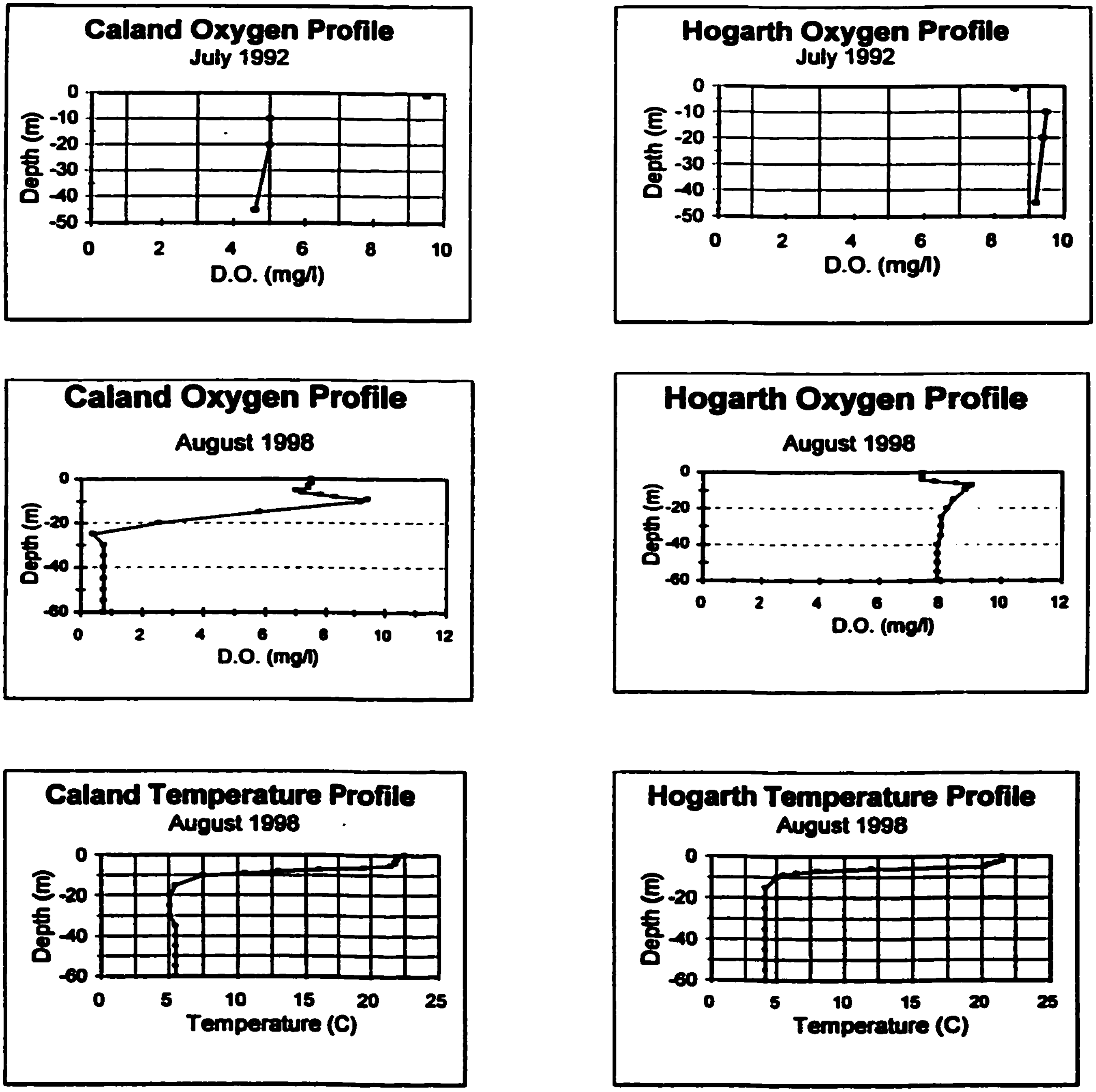


Figure 3. Dissolved oxygen and temperature profiles in Caland and Hogarth Pits.

period of time. Losses of nitrogen occur by reduction of NO_3 to N_2 gas by bacteria with subsequent return of N_2 to the atmosphere (Wetzel, 1983). In the Caland pit though, monimolimnetic levels of NO_3 have increased which may be caused by insufficient numbers or inhibition of denitrification bacteria due to high concentrations of other elements.

Oxygen

Figure 3 shows that Caland's oxygen concentration has become severely depleted below 20m depth. Axler et al.(1996) states that in intensive aquaculture systems, organic enrichment comes primarily from uneaten food and fish feces which is dispersed to the surrounding water, and numerous studies have demonstrated hypolimnetic and sediment anoxia due to this organic enrichment. However, it is uncertain at this time if this depletion is the single result of organic enrichment, oxidation to SO_4 and NO_3, or the onset of the meromictic environment as depth:surface area ratio increases.

The Caland site is currently conducive to the description of a meromictic system with its top 20m of water seasonally mixing and stratifying while its lower waters are stagnant, anaerobic, and more saline (Table 1). Caland's 1998 oxygen profile (Fig 3) as well as its temperature profile (Fig 3) are now reflective of this environment. Hogarth, however, does not fit the norm as this waterbody is oxygenated from top to bottom (Fig 3), has higher concentrations of most elements compared to Caland, and appears devoid of aquatic life.

Sulfate

In Caland, SO_4 levels have substantially decreased in the upper waters (by approx 75mg/l) and conversely increased in the lower waters (by more than 100mg/l) compared to MOE data (Table 1). These changes are most likely due to the recent formation of the meromictic environment. Reduction of sulfate to sulfide occurs under anoxic conditions by sulfate reducing bacteria. Cole (1994) states that the monimolimnia may contain appreciable quantities of the very soluble gas H_2S especially in regions high in edaphic sulfate. This gas may be released into the air or re-oxidized to SO_4 in the overlying oxic water column. This re-oxidation increases COD thereby depleting dissolved oxygen. The presence of high concentrations of SO_4 in the anoxic waters of Caland coincides with a theory that there may be too much SO_4 loading for reducing bacteria to reduce to H_2S. A potential problem at the fish farm may be elevated H_2S concentrations which are poisonous to aerobic organisms, by inactivating the enzyme cytochrome oxidase (Cole, 1994).

Mortimer (1941) suggest that anoxic sediments rich in organic matter release gaseous hydrogen sulfide. This rotten-egg odor is apparent in the sediment samples below the fish farm, as is the characteristic sapropel.

There are currently substantially higher concentrations of SO_4 in Hogarth compared to Caland and compared to prior MOE data (Table 1). Since Hogarth is oxygenated for its entire water column,SO_4 remains in its most stable form and its concentration changes little with depth. These elevated concentrations may be due to higher quantities of pyritic hematite in the Middle Arm compared to the East Arm (Ontario Department of Mines and Northern Affairs Map, 1972). The formation of acidic effluents is associated with ore bodies containing the more readily oxidizable iron sulfides, such as pyrite and pyrrhotite (Mining, Mineral and Metallurgical Processes Division, Environment Canada, 1987). Monitored (but unreported) direct drainage into Hogarth is more acidic ie. pH range of 2-5 compared to drainage into Caland pH of 7-8. This lower pH increases leaching from the waste rock and tailings and thus increases concentrations of elements in the water column.

With both pits being closed basins, salinity may increase over time (detectable in Hogarth when comparing concentration over time) depending on drainage/rainfall, solute load, evaporation, and precipitation of elements.

Calcium & Magnesium

Hogarth and Caland are buffered from acid mine drainage due to naturally occuring limestone and dolomite deposits. In Hogarth, the calcium and magnesium concentrations are 337mg/l and 208mg/l respectively, which are much higher than Caland (Table 1). Normally, the concentration in limestone/carbonate area lakes ranges for calcium from 30-100mg/L and magnesium from 5-50mg/l (Lind, 1985). These higher concentrations in Hogarth are likely due to dolomite solubilization and the release of Ca and Mg in equal proportions. Dolomite is found in higher quantities around the Hogarth minesite compared to the Caland site, which seems to have more calcium carbonate (Geological Map, 1968 & 1969). In a lake environment, as soon as the $CaCO_3$ threshold is reached with warming epilimnetic temperatures, calcite precipitates reducing the Ca:Mg ratio (Deckker and Last, 1988). Magnesium is required in the chlorophyll molecule but very high levels of magnesium salt produce anesthesia in both invertebrates and vertebrates (Cole, 1994).

Biotic Activity in Caland versus Hogarth

From plankton tows, Hogarth appeared devoid of aquatic organisms even though high concentration of oxygen extended down to the sediment/water interface (>150m). Possible explainations for the lack of life include: elevated magnesium salt concentrations producing anesthesia in invertebrates and vertebrates; increasing salinity affecting/disrupting organisms' osmotic capabilities; and/or presence of toxic substances. Ripley et al., (1996) states that one of many minor constituents commonly found in iron ores is arsenic. High levels of arsenic were found in lichen and conifer needle tissue from reports by the Hydro Generating Station on Marmion Lake. Another possible explanation may be the presence of buried toxicants (eg. PCB's) left behind by Steep Rock Inc. after closure.

Differences between these two pits can be attributed to a variety of factors: presence of the fish farm, geological variations, and, different types and/or different quantities of waste rock at the two mine sites. Changes over time, however, can be attributed to the depletion of oxygen (in the Caland site only), rising water levels submersing new rock/tailing surfaces, and increases in salinity resulting from evaporation in closed basins. Further water sampling, toxicity testing, controlled in-lab experiments, and sediment and rock analysis will be studied to determine the magnitude of effect on these environments.

REFERENCES

Axler, R., Yokom, S., Tikkanen, C., Henneck, J., M. McDonald.1995. Effects of aquaculture on mine plit lakes near Chisholm, MN: restoration of Twin City-South pitlake by fallowing and status of Fraser pit lake. Technical Report: NRRI/TR-95-04. 21+pp

Axler, R., Larsen, C., Tikkanen, C., McDonald, M., Yokom, S. and P. Aas. 1996a. Water quality issues associated with aquaculture: A case study in mine pit lakes. Water Environ. Res. 68:995-1011.

Axler, R., Henneck, J., Bridgham, S., Tikkanen, C., Nordman, D., Bamford, A., and M. McDonald. 1996b. Constructed wetlands in Northern Minnesota for treatment of aquacultrue wastes. Presented at: Constructed Wetlands in Cold Climates. Niagara-on-the-Lake, Ontario, Canada

Bergheim, A., Aabel, J.P., and E.A. Seymour. 1991. Past and present approaches to aquaculture waste management in Norwegian net pen culture operations. *in* C.B. Cowey and C.Y. Cho (eds.) Nutritional Strategies in Management of Aquaculture Waste. Proc. First International symposium on Nutritional Strategies in Management of Aquaculture Waste. 5-8 June 1990, Guelph, Ontario, CAN. pp117-136.

Chapman, G. 1997. Assessment of the Biological Capacity of Caland Lake. Unpublished Manuscript Prepared for Snow Lake Fish Farms. 16pp

Cloud, P. 1973. Paleoecological significance of the banded iron-formation. Econ. Geol. 68:1135-1143

Cole, G.A. 1994. Textbook of Limnology 4th ed. Waveland Press Inc: Illinois. 412pp

Cornel, G.E., and F.G. Whoriskey. 1993. The effects of rainbow trout (*Oncorhynchus mykiss*) cage culture on the water quality, zooplankton, benthos and sediments of Lac du Passage, Quebec. Aquaculture 109:101-117

DeDeckker, P., and W.M. Last. 1988. Modern dolomite deposition in continental, saline lakes, western Victoria, Australia. Geology. 16:29-32.

Eugster, H.P. and L.A. Hardie. *in* A. Lerman. 1978. Lakes: Chemistry, Geology, Physics. Springer-Verlag: New York. p237-293.

Geological Map, 1968 & 1969 *in* Shklanka, R. 1972. Geology of the Steep Rock Lake area district of Rainy River. Ont. Dept. Mines and Northern Affairs. Geol. Rep. 93. 114pp.

Lakehead University Environmental Lab. 1998. Standard Operating Procedures.

Lind, O.T. 1985. Handbook of common methods in Limnology. 2nd ed. Kendall/Hunt Pub. Co: Iowa. 199pp

Mining, Mineral and Metallurgical Processes Division. 1987. Mine and Mill Wastewater Treatment. EPS 2/MM/3 Environment Canada. 86pp

Mortimer, C.H. 1941. The exchange of dissolved substances between mud and water in lakes. (parts I & II). J. Ecol. 29:280-329.

Ojakangas, R.W. and C.L. Matsch. 1982. Minnesota's Geology. University of Minn. Press. 255p.

Ontario Department of Mines and Northern Affairs. Map 2217. 1970. Steep Rock Lake Area, Rainy River District

Ontario Ministry of Environment and Energy. 1988-1993. Unpublished data. Thunder Bay, Canada

Ontario Ministry of Northern Development and Mines. 1994. Rock Ontario.Ontario Ministry of Northern Development and Mines. Rock On Series 1, 89p.

Pye, E.G. 1968. Geology and scenery, Rainy Lake and east to Lake Superior. Ont. Dept. Mines, Geol. Guide Book 1, 114pp.

Shklanka, R. 1972. Geology of the Steep Rock Lake area district of Rainy River. Ont. Dept. Mines and Northern Affairs. Geol. Rep. 93. 114pp.

Snow Lake Fish Farms. 1997. Snow Lake fish farms business plan. Unpublished Manuscript. 41pp.

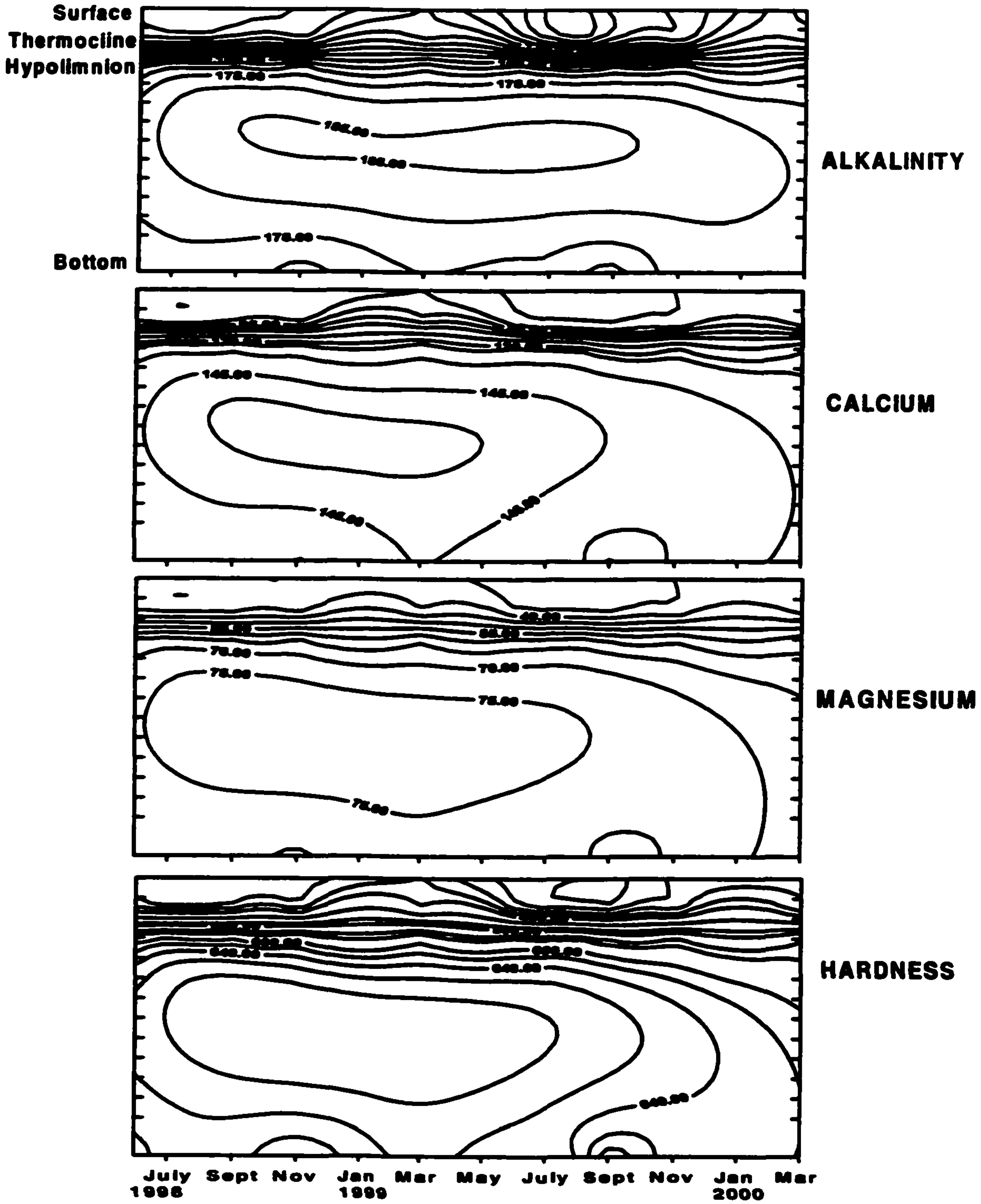
Steep Rock Mines. 1943. The Steep Rock iron ore deposits. A story of patient investigation and development. Can. Mining. J. 63:437-444.

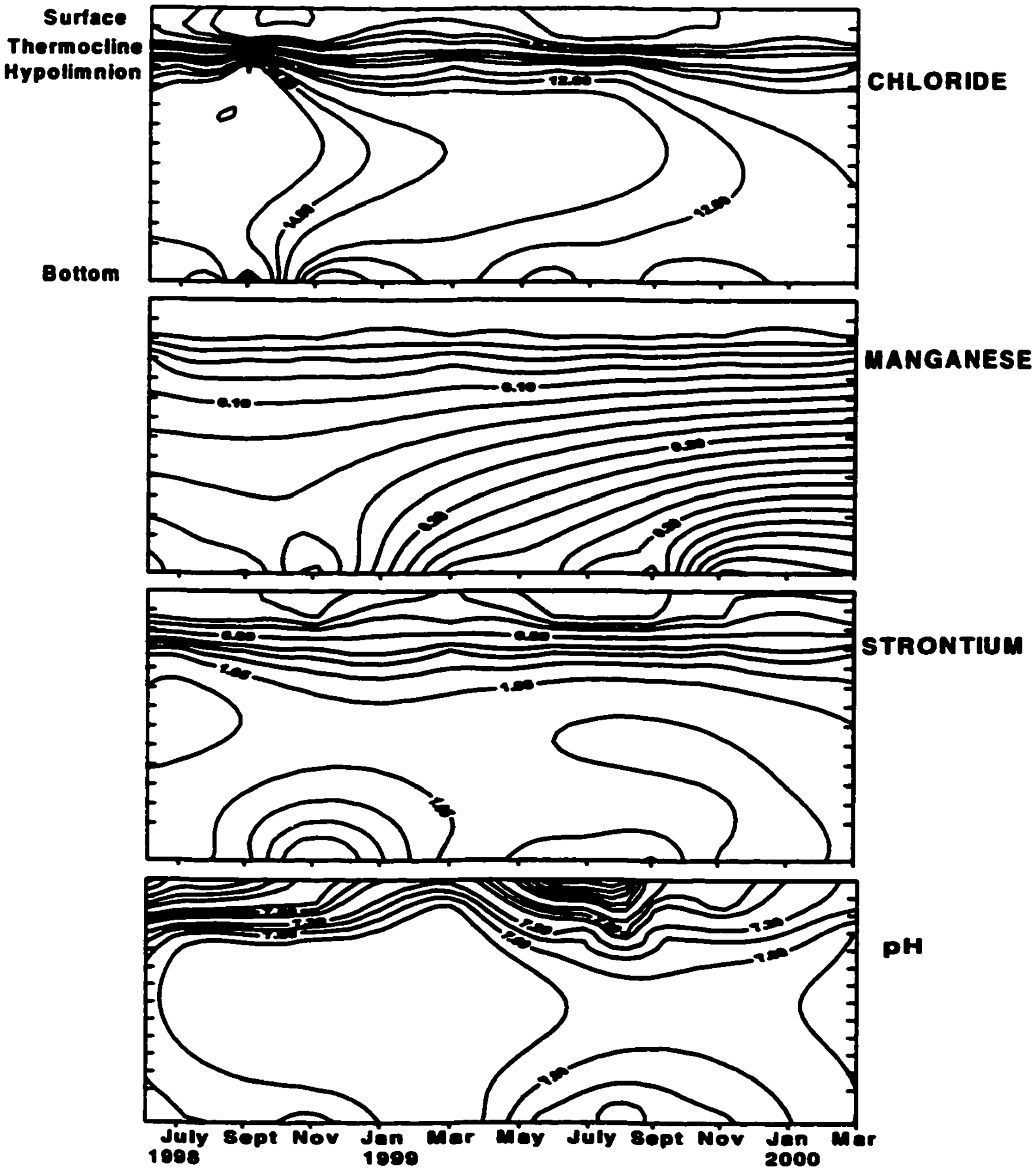
Ripley, E.A., Redmann, R.E., A.A. Crowder. 1996. Environmental Effects of Mining. St. Lucie Press: Florida. 356pp

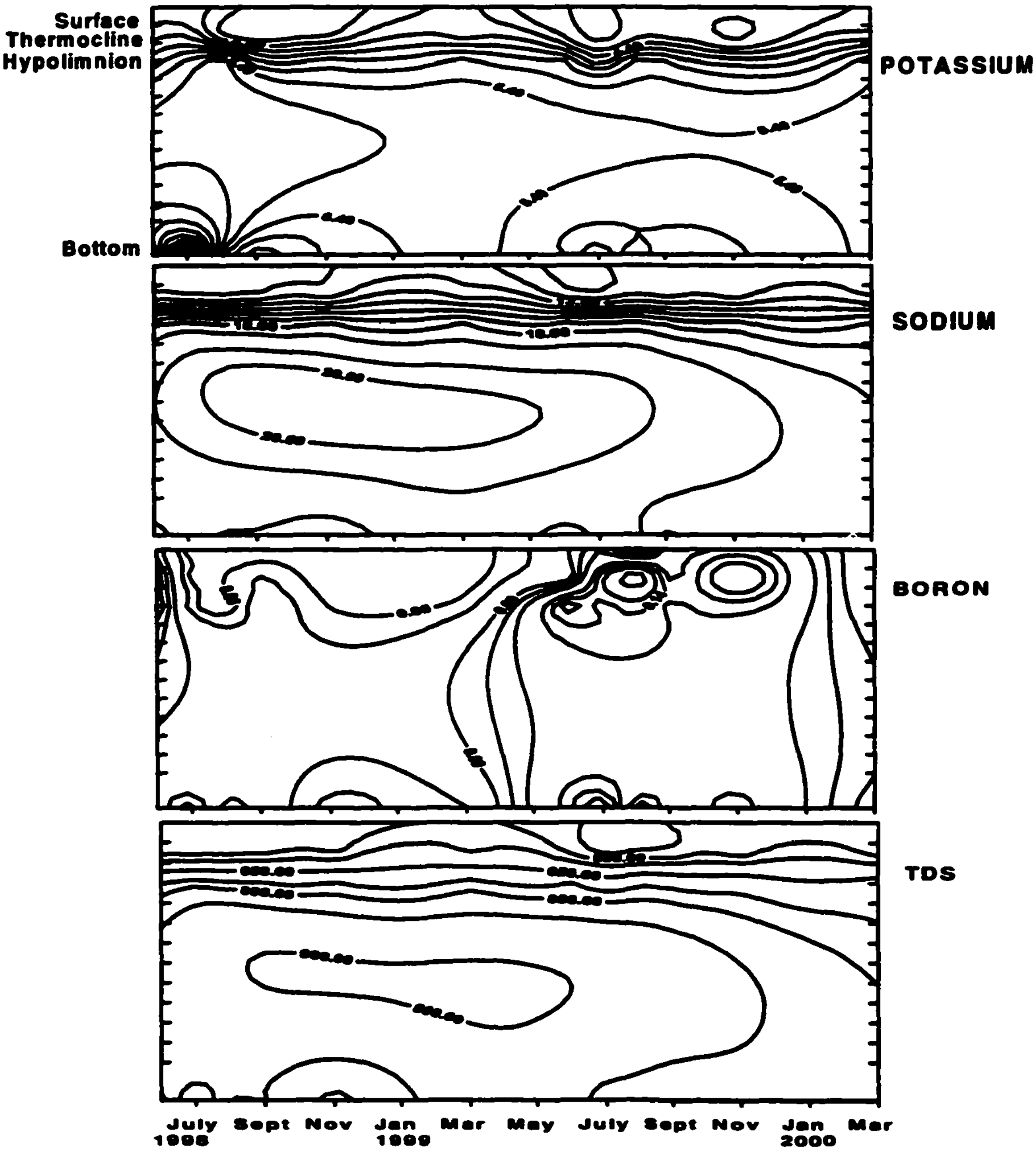
Wetzel, R.G. 1983. Limnology 2nd ed. Saunders College Publishing: Toronto, Ontario. 767pp.

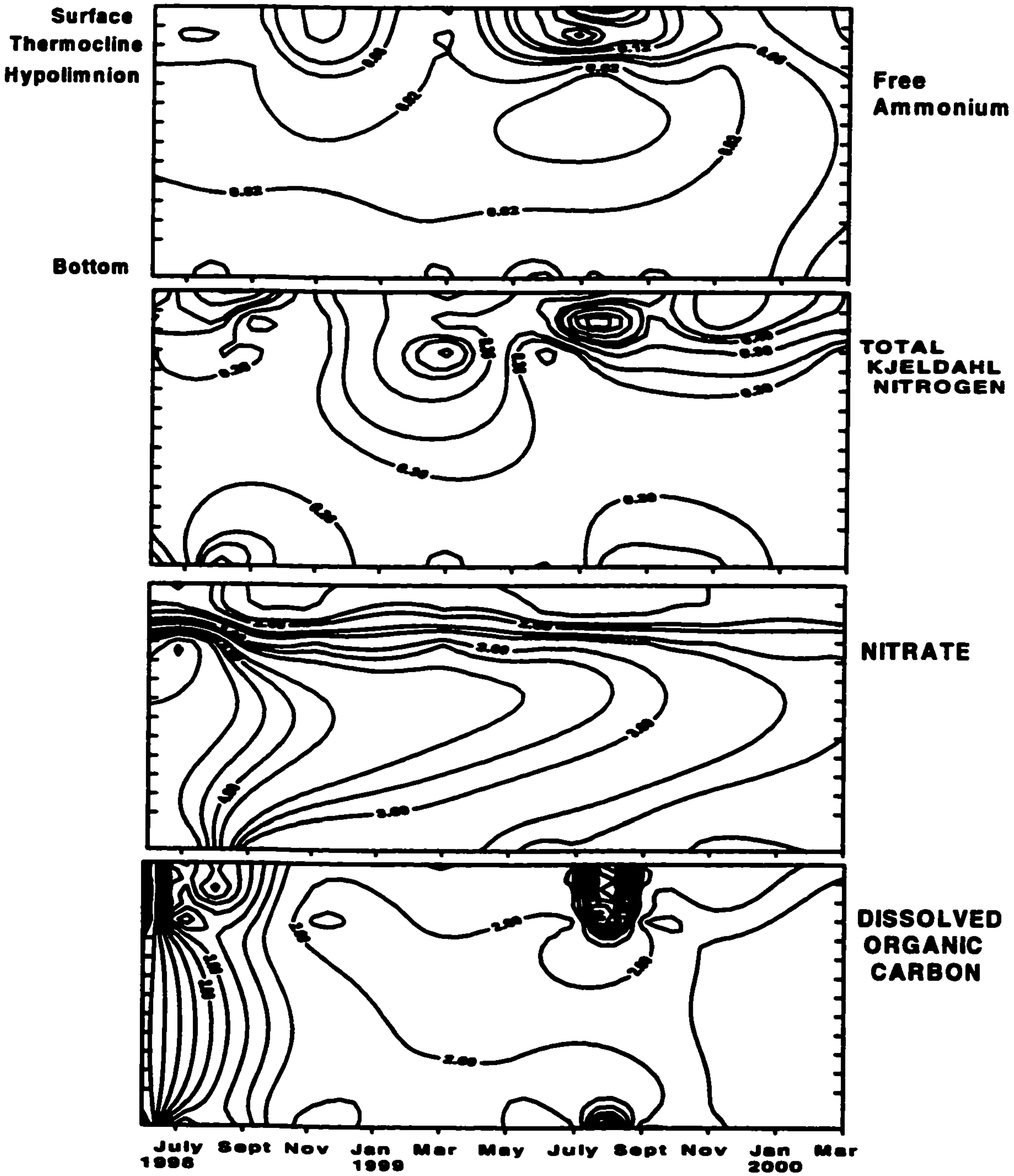
Appendix B

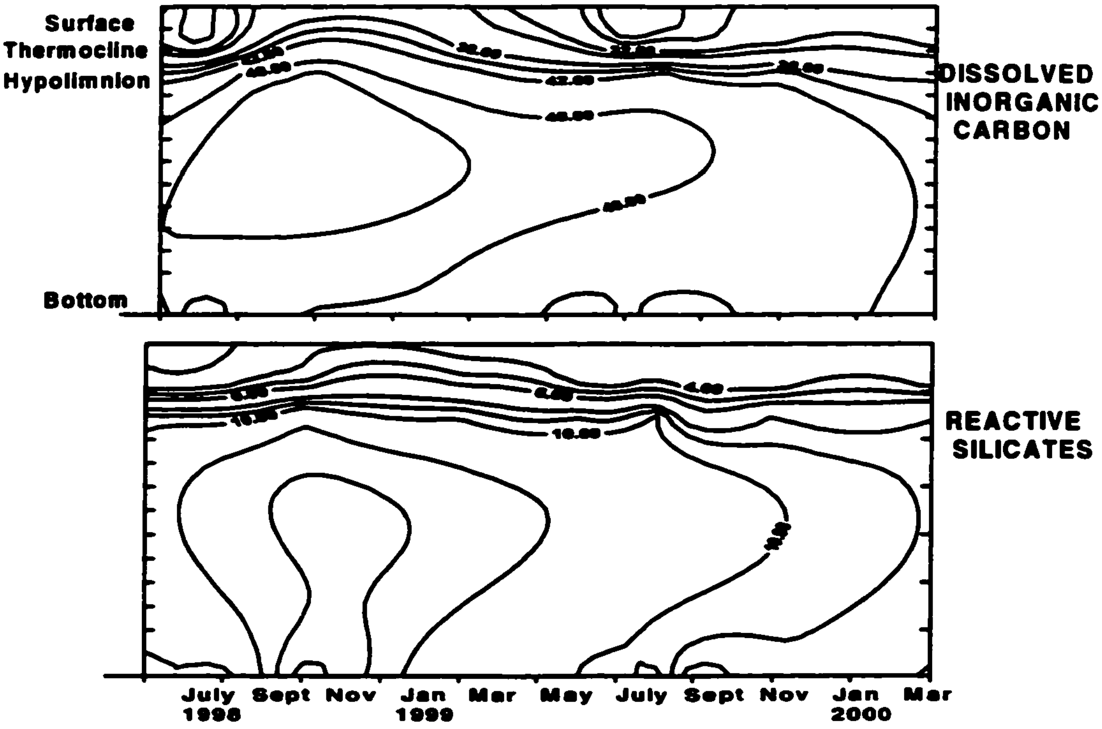
The following parameters are average concentrations of the three sites at represented depths shown for each sampling occasion in Caland pit-lake. Each of the parameters are in $mg \cdot l^{-1}$ except alkalinity ($mgCaCO_3 \cdot l^{-1}$), hardness (calc), and pH.


Surface
Thermocline
Hypolimnion
Bottom
175.00
185.00
ALKALINITY
145.00
CALCIUM
40.00
60.00
70.00
75.00
MAGNESIUM
640.00
HARDNESS
July 1998
Sept
Nov
Jan 1999
Mar
May
July
Sept
Nov
Jan 2000
Mar


Surface
Thermocline
Hypolimnion
Bottom
CHLORIDE
12.00
12.00
MANGANESE
0.10
0.10
STRONTIUM
pH
July 1998
Sept
Nov
Jan 1999
Mar
May
July
Sept
Nov
Jan 2000
Mar

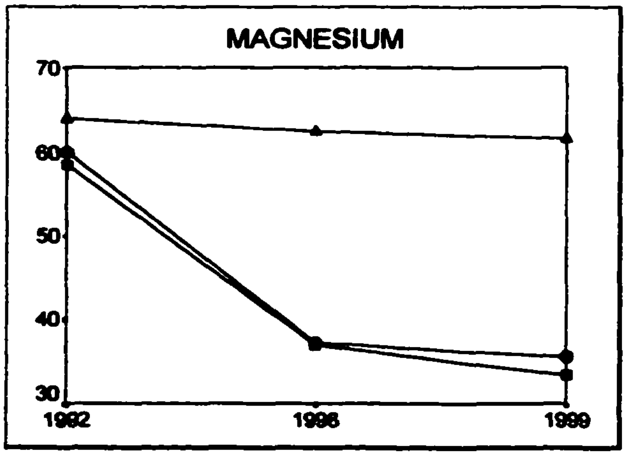

Surface
Thermocline
Hypolimnion
Bottom
POTASSIUM
SODIUM
BORON
TDS
July 1998
Sept
Nov
Jan 1999
Mar
May
July
Sept
Nov
Jan 2000
Mar


Surface
Thermocline
Hypolimnion
Bottom
Free Ammonium
TOTAL KJELDAHL NITROGEN
NITRATE
DISSOLVED ORGANIC CARBON
July 1998
Sept
Nov
Jan 1999
Mar
May
July
Sept
Nov
Jan 2000
Mar


Surface
Thermocline
Hypolimnion
Bottom
DISSOLVED INORGANIC CARBON
38.00
42.00
46.00
REACTIVE SILICATES
4.00
10.00
July 1998
Sept
Nov
Jan 1999
Mar
May
July
Sept
Nov
Jan 2000
Mar

Appendix - C

A water chemistry comparison between data collected in June 1992, 1998 and 1999 in Caland pit-lake, Ontario. The 1992 data was collected by OMOE. Represented depths are: epilimnion (▪), thermocline (•), and hypolimnion (▴). Concentrations are in $mg \cdot l^{-1}$ except for alkalinity ($mgCaCO_3 \cdot l^{-1}$) and conductivity ($uS \cdot cm^{-1}$).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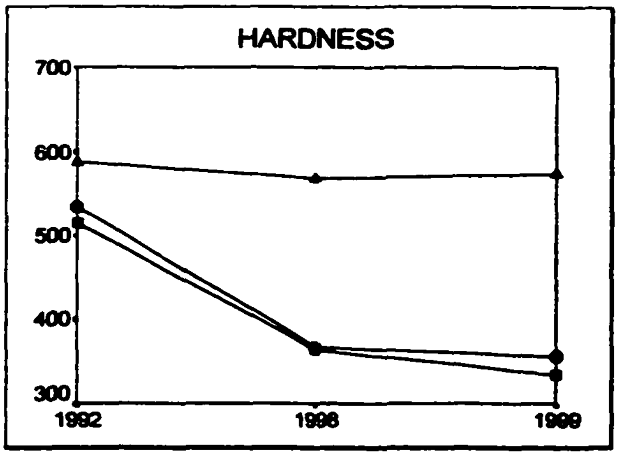
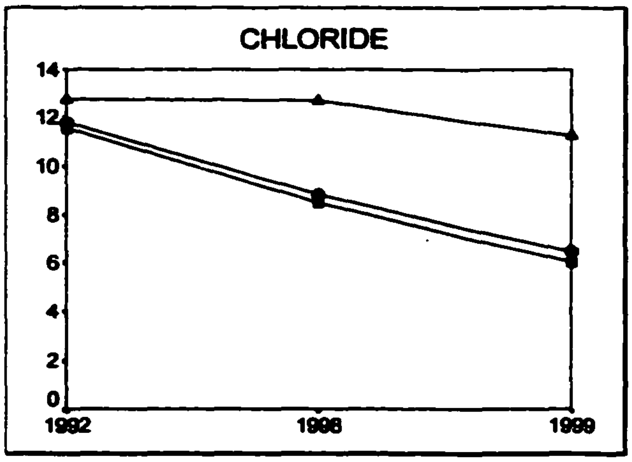
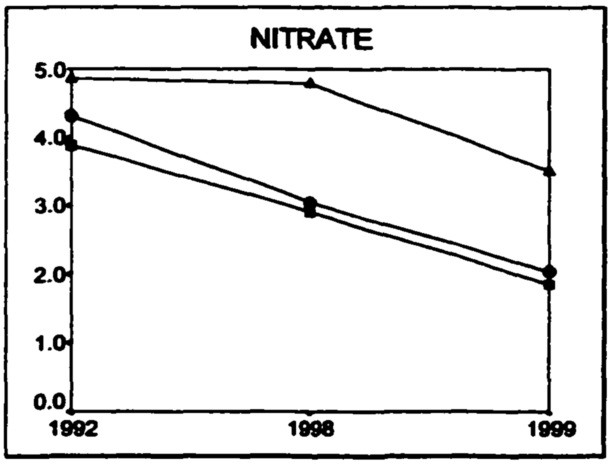

HARDNESS


CHLORIDE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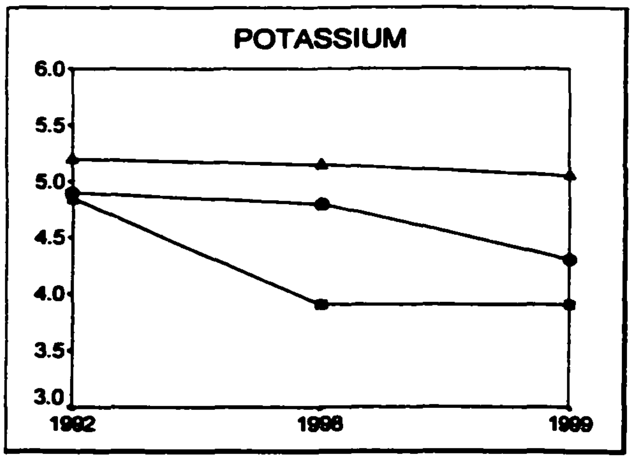
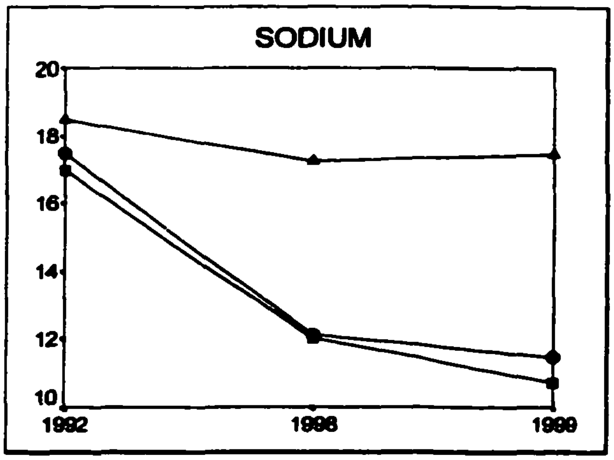
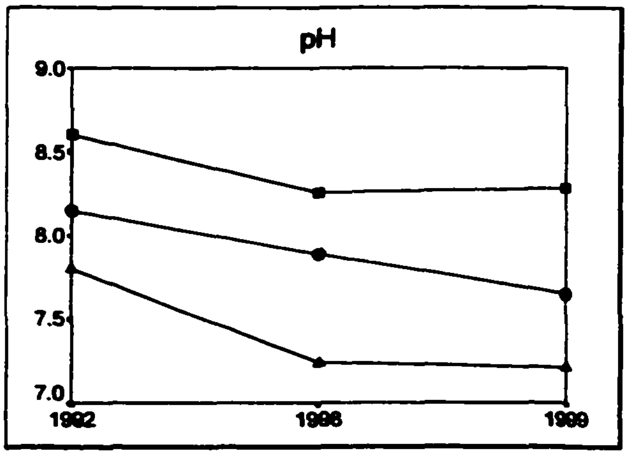


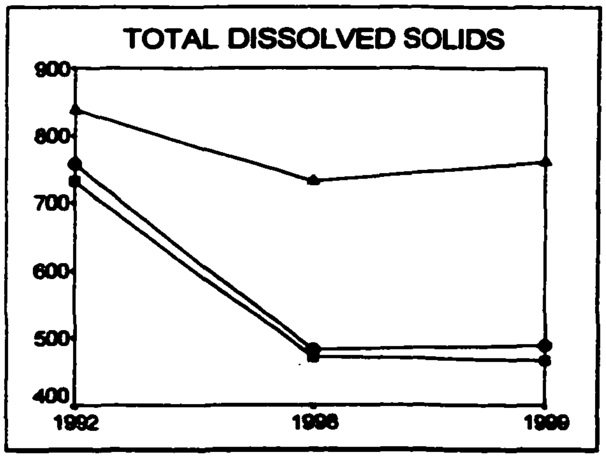
POTASSIUM



SODIUM



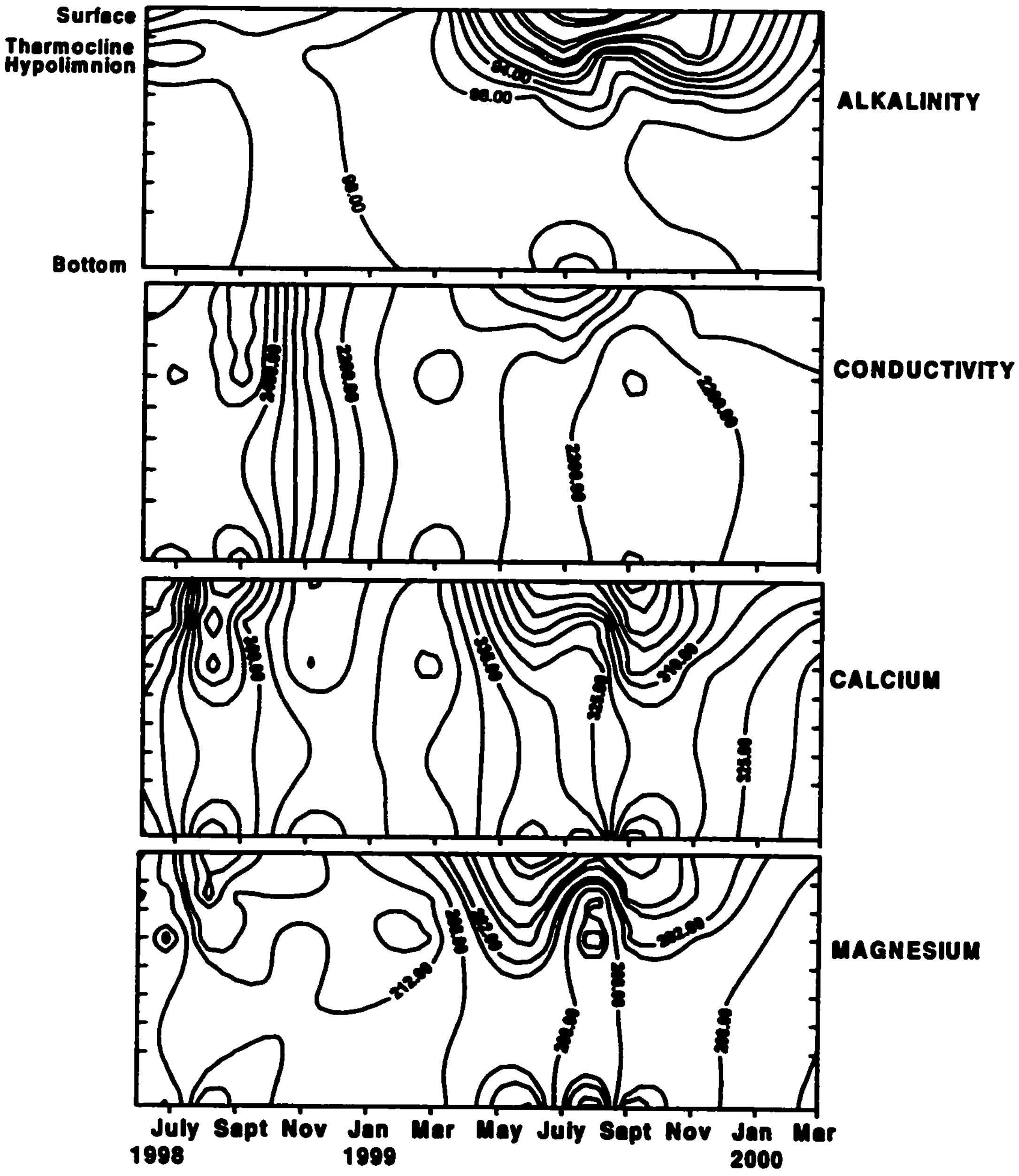
pH

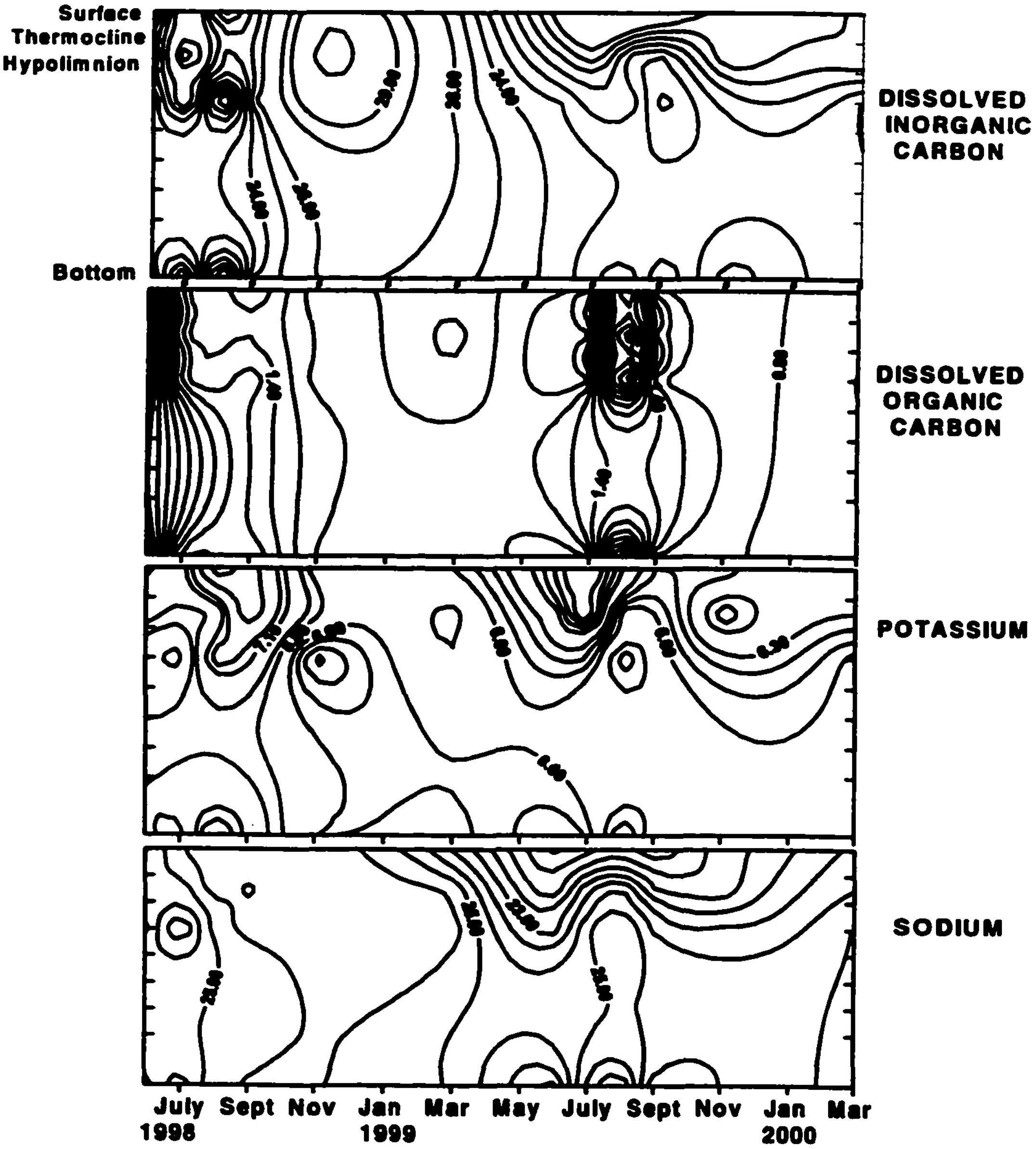


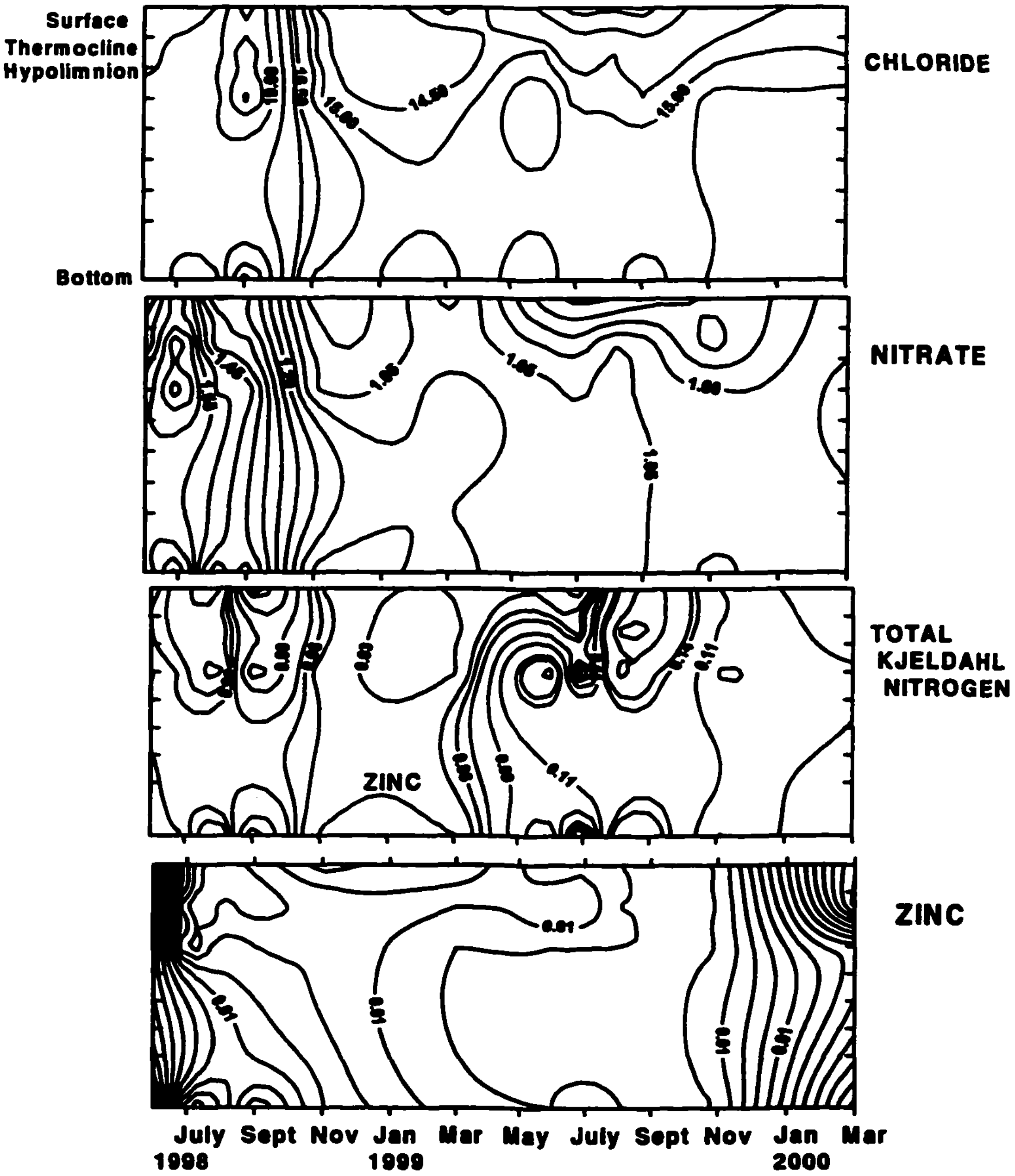
TOTAL DISSOLVED SOLIDS

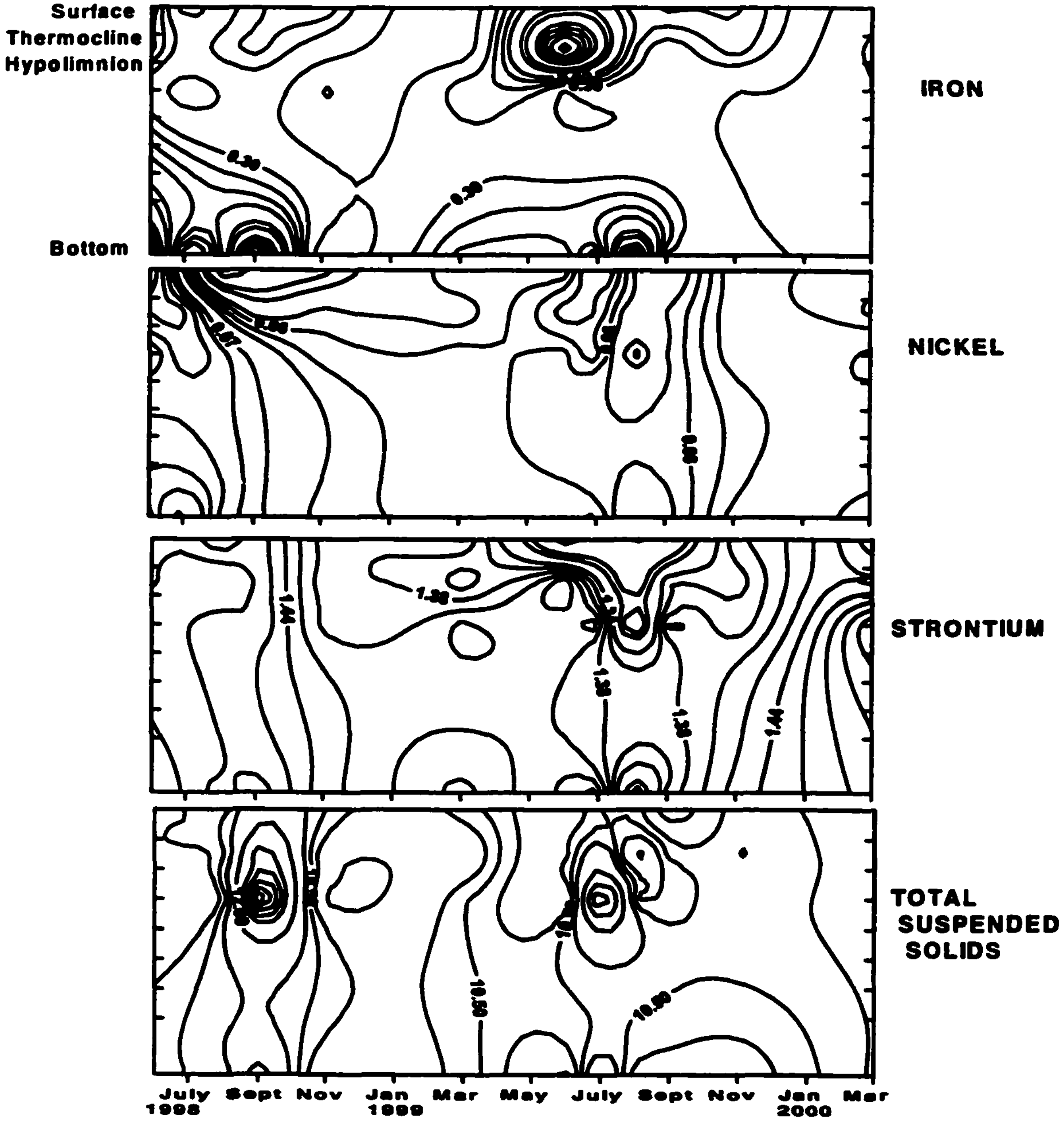
Appendix D

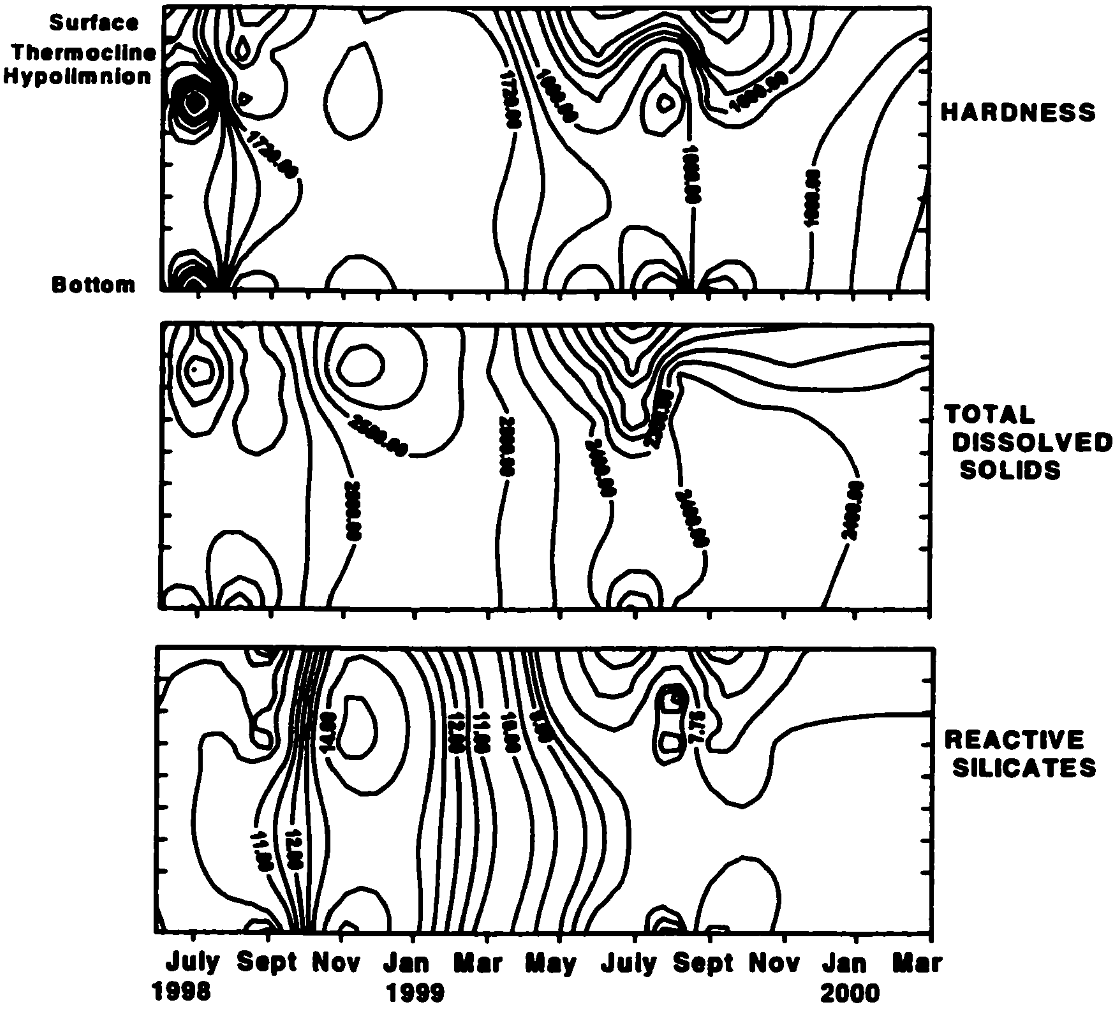
The following parameters are average concentrations of the three sites at represented depths shown for each sampling occasion in Hogarth pit-lake. Each of the parameters are in mg·l^{-1} except alkalinity (mgCaCO$_3$·l^{-1}), hardness (calc), and pH.


Surface
Thermocline
Hypolimnion
Bottom
ALKALINITY
CONDUCTIVITY
CALCIUM
MAGNESIUM
July
1998
Sept
Nov
Jan
1999
Mar
May
July
Sept
Nov
Jan
2000
Mar


Surface
Thermocline
Hypolimnion
Bottom
DISSOLVED INORGANIC CARBON
24.00
26.00
DISSOLVED ORGANIC CARBON
1.40
POTASSIUM
7.10
6.30
SODIUM
July 1998
Sept
Nov
Jan 1999
Mar
May
July
Sept
Nov
Jan 2000
Mar


Surface
Thermocline
Hypolimnion
Bottom
CHLORIDE
14.50
NITRATE
1.86
TOTAL KJELDAHL NITROGEN
0.11
ZINC
ZINC
0.01
July 1998
Sept
Nov
Jan 1999
Mar
May
July
Sept
Nov
Jan 2000
Mar


Surface
Thermocline
Hypolimnion
Bottom
IRON
NICKEL
STRONTIUM
TOTAL SUSPENDED SOLIDS
0.30
1.38
1.44
10.50
July 1998
Sept
Nov
Jan 1999
Mar
May
July
Sept
Nov
Jan 2000
Mar


Surface
Thermocline
Hypolimnion
Bottom
HARDNESS
TOTAL DISSOLVED SOLIDS
REACTIVE SILICATES
July 1998
Sept
Nov
Jan 1999
Mar
May
July
Sept
Nov
Jan 2000
Mar