INFORMATION TO USERS

This manuscript has been reproduced from the microfilm master. UMI films the text directly from the original or copy submitted. Thus, some thesis and dissertation copies are in typewriter face, while others may be from any type of computer printer.

The quality of this reproduction is dependent upon the quality of the copy submitted. Broken or indistinct print, colored or poor quality illustrations and photographs, print bleedthrough, substandard margins, and improper alignment can adversely affect reproduction.

In the unlikely event that the author did not send UMI a complete manuscript and there are missing pages, these will be noted. Also, if unauthorized copyright material had to be removed, a note will indicate the deletion.

Oversize materials (e.g., maps, drawings, charts) are reproduced by sectioning the original, beginning at the upper left-hand corner and continuing from left to right in equal sections with small overlaps.

Photographs included in the original manuscript have been reproduced xerographically in this copy. Higher quality 6" x 9" black and white photographic prints are available for any photographs or illustrations appearing in this copy for an additional charge. Contact UMI directly to order.

Bell & Howell Information and Learning
300 North Zeeb Road, Ann Arbor, MI 48106-1346 USA
800-521-0600

UMI®

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

THE PREDICTION OF A MODERN GOLD MINE'S CONTAMINANT DISPERSION AND DEPOSITION IN A RIVER MIXING ZONE

by

DERRICK A. KWIATKOWSKI H.B.Sc.

A thesis presented to Lakehead University in partial
fulfillment of the requirements for the degree of M.Sc. In the
Department of Biology.

Thunder Bay, Ontario

National Library
of Canada

Acquisitions and
Bibliographic Services

395 Wellington Street
Ottawa ON K1A 0N4
Canada

Bibliothèque nationale
du Canada

Acquisitions et
services bibliographiques

395, rue Wellington
Ottawa ON K1A 0N4
Canada

Your file Votre référence

Our file Notre référence

The author has granted a non-exclusive licence allowing the National Library of Canada to reproduce, loan, distribute or sell copies of this thesis in microform, paper or electronic formats.

L'auteur a accordé une licence non exclusive permettant à la Bibliothèque nationale du Canada de reproduire, prêter, distribuer ou vendre des copies de cette thèse sous la forme de microfiche/film, de reproduction sur papier ou sur format électronique.

The author retains ownership of the copyright in this thesis. Neither the thesis nor substantial extracts from it may be printed or otherwise reproduced without the author's permission.

L'auteur conserve la propriété du droit d'auteur qui protège cette thèse. Ni la thèse ni des extraits substantiels de celle-ci ne doivent être imprimés ou autrement reproduits sans son autorisation.

0-612-54512-1

Canada

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.

The prediction of a modern gold mine's contaminant dispersion and deposition in a river mixing zone.

Abstract:

A predictive spatial model of contaminant dispersion and deposition was produced for the river mixing zone of Placer Dome Canada's Musselwhite Mine. Discriminant analysis classified water and sediment quality into three groups based upon seven parameters; total aluminum, total zinc, total copper, total iron, nitrate, sulphate, and pH at three depths. The first group consisted of three sites situated immediately outside of the mine's outfall, and was characterized by high mean discriminant function scores used to describe both water and sediment. The second and third groups consisted of the remainder of sites downstream and the control sites respectively, and were characterized by relatively lower scores in the water and sediment. Spatial predictions of contaminant dispersion and deposition, based upon the regression of discriminant function score to effluent tracer measurements taken at sampling sites, delineated an area of lower water and sediment quality within the vicinity of outfall. The effluent-receiving water body, within a minimum mixing length of 300 m, was found to be capable of diluting all parameters identified as being significantly higher at point of outfall to concentrations that were not significantly different from those found upstream of discharge. Parameters of current or potential environmental concern in the mine's effluent mixing zone were also identified in this study.

Introduction:

Modern gold mines often employ a number of environmental control systems to recover, remove, or destroy contaminants such as cyanide, dissolved metals, and nitrogenous species that may be present in their final effluent. These systems include; a tailings washing circuit for recovery of cyanide used during gold extraction, the use of specialized treatment processes such as the INCO SO2/air treatment system for the removal of cyanides and dissolved metals from tailings slurry, the natural degradation and sedimentation of contaminants in submerged tailings impoundments, and the use of polishing ponds and wetlands for final effluent treatment. Once treated, the contaminants are further diluted when discharged into the receiving environment.

Contaminant Sources:

The potential sources of contamination in modern gold mine effluent include the following;

1. Acid Mine Drainage

Acid mine drainage, or AMD, is considered to be the largest single environmental problem facing the mineral industry in Canada (Ferguson & Erikson, 1987), and has been defined as a low pH leachate product that is formed by the natural oxidation of common iron-sulphur minerals such as pyrite (FeS_2) and pyrrhotite ($Fe_{1-x}S$), and has been reported to develop in three stages.

First stage AMD occurs at a pH above 4.5, in which pyrite, pyrrohotite, and other sulfide minerals are slowly and abiotically oxidized to produce sulphuric acid (H_2SO_4), and ferrous sulphate ($FeSO_4$) (equations 1 & 2). In the presence of oxygen, the ferrous iron produced in (1) and (2) is further converted to ferric iron (equation 3) to yield ferric sulphate ($Fe_2(SO_4)_3$), while the sulphuric acid generated may be neutralized by carbonate containing minerals such as calcite ($CaCO_3$) or dolomite ($CaMgCO_3$) if present to form gypsum ($CaSO_4 \bullet 2H_2O$) (Equations 4 & 5).

First stage AMD reactions;

Pyrite oxidation;

$$FeS_2 + 7/2\ O_2 + H_2O \rightarrow Fe^{2+} + 2SO_4^{2-} + 2\ H^+ \qquad (1)$$

Pyrrohotite oxidation;

$$Fe_{1-x}S(s) + (2-x/2)\ O_2 + xH_2O \rightarrow (1-x)Fe^{2+} + SO_4^{2-} + 2xH^+ \qquad (2)$$

Oxidation of ferrous Iron;

$$Fe^{2+} + 1/4\ O_2 + H^+ \rightarrow Fe^{3+} + \tfrac{1}{2}\ H_2O \qquad (3)$$

Possible carbonate buffering systems for First Stage AMD;

$$CaCO_3 + H_2SO_4 \rightarrow CaSO_4 + H_2O + CO_2 \qquad (4)$$

$$CaMg(CO_3)_2 + 2H_2SO_4 \rightarrow CaSO_4 + MgSO_4 + 2H_2O + 2CO_2 \qquad (5)$$

Second stage AMD development is characterized by a depletion of carbonate buffering mineral reserves, followed by a drop in the aqueous, microenvironmental pH surrounding the sulphide bearing minerals. As the pH drops, site specific strains of *Thibacilus ferroxidans* begin to multiply and catalyze the oxidation of the sulphidic minerals at rates of six orders of magnitude greater than chemical oxidation alone (Evangelou & Zharng, 1995). This increased bacterial activity results in the further decrease in pH until the third stage of AMD is reached at a pH of 3.

During third stage AMD, the ferric ion remains in solution, and begins to liberate additional metal ions through oxidation as is shown in equation (6), simplified in equation (7).

Third Stage AMD reactions propagated by aqueous ferric ions;

$$MeS + Fe_2(SO_4)_3 + 3/2O_2 + H_2O \rightarrow MeSO_4 + 2FeSO_4 + H_2SO_4 \qquad (6)$$

$$2Fe^{3+} + MeS \longrightarrow 2Fe^{2+} + Me^{+} + S \qquad (7)$$

(Where: MeS = solid metal sulphide; Fe^{3+} = aqueous ferric iron ion; Me^{+} = aqueous metal ion; S = sulphur; Fe^{2+} = aqueous ferrous iron ion.)

Acid mine drainage can therefore create two distinct environmental problems for modern mining operations; 1) the acidification of waters draining from exposed rock surfaces containing sulphidic minerals, and 2) the solubilization of metals caused by ferric ion oxidation. Typically, effluent affected by AMD will show the following characteristics; a pH of between 2 to 4 associated with sulphuric acid generation, concentrations of dissolved iron typically in the range of 100-3000 mg/L, elevated (1-200 mg/L) concentrations of other dissolved metals such as copper, zinc, and lead, and high concentrations of dissolved salts including calcium, magnesium, aluminum, and sulphate (100-30 000 mg/L) (Ritchie, 1994).

2. Byproducts of the Milling Process

In many modern mines, gold is first extracted from ore though leaching in which cyanide is introduced under alkaline conditions to dissolve the precious metal from the host rock. This gold dissolution process, better known as cyanidation, can be summarized by Elsner's equation (8) (Smith & Mudder, 1991);

Elsner's equation;

$$4Au + 8NaCN + O_2 + 2H_2O \rightleftharpoons 4NaAu(CN)_2 + 4NaOH \qquad (8)$$

During cyanidation, metallic and non-metallic species may become liberated from the ore to compete with gold for cyanide by forming a wide range of dissolved cyanide complexes. The major components of ore that commonly form such complexes include copper, iron, arsenic, zinc, and sulphide minerals. Of these, sulphides can be particularly effective in inhibiting cyanidation during conditions of warm slurry temperature (Personal Communication, Musslewhite), or during the processing of exceptionally sulphide rich ores (Smith and Mudder, 1991). The soluble products of sulphide mineral decomposition, namely polysulphides (i.e., CaS_5) and thiosulphates (i.e., CaS_2O_3), accomplish this by combining with cyanide to form thiocyanate (CNS^-), and by consuming the oxygen required in Elsner's reaction according to the following reactions (9 and 10);

$$2CaS_5 + O_2 + 10NaCN + 2H_2O \rightarrow 10NaCNS + 2CaS_2O_3 \qquad (9)$$

$$CaS_2O_3 + NaCN + 1/2O_2 + 2H_20 \rightarrow CaSO_4{\bullet}2H_20 + NaCNS \qquad (10)$$

Gold that has been extracted through cyanidation is commonly absorbed from solution onto activated carbon through techniques such as the Carbon in Pulp (CIP) process, and is then recovered through further processing. Untreated waste from cyanidation and CIP processing typically consists of a slurry with elevated levels of dissolved free cyanide, metallo-cyanide complexes, cyanates, thiocyanates, and other cyanide related compounds (Higgs et al., 1992), and must often undergo additional treatment before being released as final mine effluent.

The ability to effectively remove or lower concentrations of cyanide related contaminants in process effluent is a primary concern of the Canadian gold mining industry. Modern treatment methods designed for cyanide removal that are currently used in Canada include; natural degradation, alkaline chlorination, the Noranda SO_2 process, and the Inco SO_2/Air process, (Scott, 1987). The neutralization of free acid and removal of dissolved metals from effluent is commonly accomplished through metal hydroxide precipitation under alkaline conditions at many Canadian mines.

3. Groundwater contamination through tailings seepage.

The use of submerged impoundments for long term storage of potentially acid generating tailings is commonly practiced in the Canadian gold mining industry (Scott, 1987). 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 (Feasby, 1993). Shallow water covers also allow for the natural degradation of cyanide from the aqueous decant primarily through the processes of ultraviolet degradation, the aging of the tailings slurry, and volatilization. Ultraviolet degradation involves the photolysis of metallo-cyanide complexes that can be brought about by exposure to sunlight to yield free cyanide ions (CN^-) in solution. The aging of the tailings slurry, in which the pH of the slurry solution is reduced due to rainfall and carbon dioxide uptake, leads to the formation of hydrocyanic acid (HCN) from these liberated cyanide ions. The volatilization of hydrocyanic acid from the aqueous environment marks the final step in the natural degradation process. Once in the atmosphere, hydrocyanic acid is believed to oxidize to carbon dioxide and ammonia via hydroxyl radicals (Howard, 1991).

The potential for mobility of dissolved contaminants into nearby surface waters through groundwater drainage is a potential environmental hazard associated with mine tailings disposal and storage. Groundwater seepage originating from gold mine tailings impoundments will typically contain a mixture of metallo-cyanide complexes, dissolved metals, thiocyanate, ammonia, and other products of natural cyanide decomposition, with seepage rates largely determined by the degree of attenuation and porosity achieved in the underlying soil.

4 Nitrogenous Blasting Residues.

The release of nitrate and ammonia into mine water or surface runoff may occur as a consequence of explosives use during blasting at a mine. An influx of such nitrogenous species into an aqueous receiving environment can result in a negative impact on water quality if contaminant loading rates exceed natural consumption processes. Both ammonia and nitrate enrichment can lead to an increase in phytoplankton productivity when growth is not limited by other nutrients such as phosphorus or by light availability (Wetzel, 1983). Ammonia, particularly when in equilibrium with its unionized state, has been shown to be capable of being a significant toxicant to the aquatic biota (Herbert and Shurben, 1965).

The INCO SO_2/Air treatment process;

The INCO SO_2/Air reactor is a single stage, continuous treatment process, capable of treating both slurry and solution for remaining total cyanide and dissolved metals present (Robbins, 1996). In the reactor, free cyanide is first oxidized to form cyanate and sulphuric acid (Equation 11). Weakly complexed metallo-cyanides are also oxidized after dissociating under the alkaline conditions encountered in the reactor (equation 12). The hydrolysis of cyanate quickly follows oxidation to form ammonium and carbonate ions (equation 13).

Cyanide oxidation reactions of the INCO SO_2/Air treatment process;

$$CN^- + SO_2 + O_2 + H_2O \rightarrow OCN^- + H_2SO_4 \qquad (11)$$

$$Me(CN)_4^{2-} + SO_2 + 4O_2 + 4H_2O \rightarrow 4OCN^- + 4H_2SO_4 + Me^{2+} \qquad (12)$$

(Where $Me^{2+} = Zn^{2+}, Cu^{2+}, Ni^{2+}, Cd^{2+}$, *etc...*)

Cyanate hydrolysis;

$$OCN^- + 2H_2O \rightarrow CO_3^{2-} + NH^{4+} \qquad (13)$$

In order for the INCO SO_2/Air treatment process to function properly, a number of chemicals including liquid sulphur dioxide, oxygen, copper sulphate, and hydrated lime must be added to the tailings slurry within the reactor. Liquid sulphur dioxide, and oxygen bubbled into solution from the bottom of the reactor, supply the oxygen and sulphur dioxide consumed in equations (11) and (12). The addition of copper sulphate supplies the slurry with dissolved copper cations that act as a catalyst in equations (11) and (12). An alkaline operational feed usually maintained within reactor through the addition of calcium hydroxide (hydrated lime) to the mix which neutralizes the sulphuric acid generated during cyanide oxidation. The product of this neutralization process, illustrated in equation (14), is a precipitate of calcium sulphate (gypsum).

Sulphuric acid neutralization through liming;

$$H_2SO_4 + Ca(OH)_2 \rightarrow CaSO_4{\bullet}2H_20\ (s) \qquad (14)$$

Liberated metal ions are precipitated out of slurry solution in the INCO SO2/Air process after reacting with calcium hydroxide to form various metal hydroxides (equation 15).

Precipitation of metal hydroxides;

$$Me^{2+} + Ca(OH)_2 \rightarrow Me(OH)_2\ (ppt) + Ca^{2+} \qquad (15)$$

Strongly complexed iron cyanides formed during cyanidation are not decomposed in the INCO SO2/Air process, but are removed instead as insoluble base metal ferrocyanide precipitates as illustrated in equation (16);

Ferrocyanide precipitation by dissolved metal cations:

$$2Me^{2+} + Fe(CN)_6^{4-} + 7H_20 \longrightarrow Me_2Fe(CN)_6{\bullet}7H_20 \text{ (ppt)} \qquad (16)$$

The additional metal often reacting with these complexes is dissolved copper which is originally added as a catalyst to promote cyanide oxidation. Dissolved copper must be maintained in a 3:1 ratio with iron in order for the precipitation of ferrocyanide to proceed at the Musselwhite mill (Personal communication). Cyanide decomposition and metal hydroxide precipitation at the mine is continuous down the length of the tailings pipe to the tailings impoundment area.

Thiocyanate oxidation within the IncoSO_2/Air reactor:

$$SCN^- + 4SO_2 + 2O_2 + 5H_2O \rightarrow OCN^- + 5H_2SO_4 \qquad (17)$$

Generally, only low quantities (less than 10 percent) of thiocyanate (SCN^-) produced from cyanidation can be oxidized to cyanate in the IncoSO2/Air treatment process (equation 7). .

Additional natural treatment strategies:

The use of artificially created wetlands has repeatedly been shown in the past to be capable of removing both cyanide (Gormely et. al.,1990), and dissolved metals from mine effluent waters. Wetlands and marshes can be capable of achieving final effluent qualities that are higher in water quality than unaffected streams (Burris et al.,1984). Wielder (1990) has reported maximum binding capacities in sphagnum peat for dissolved metals such as Al^{3+}, Fe^{2+}, Ni^{2+}, and Zn^{2+} that are nearly double that of other organic substrates, while Robbins (1985) has reported the removal of 50 to 70 percent dissolved iron from AMD affected waters after passing through *sphagnum* rich wetlands.

Biological degradation and sorption onto organic detritus have been identified as key removal processes for cyanide by wetlands plants (Higgs et. al., 1992), while it is generally believed that metal removal is carried out through the processes of adsorption through ion exchange, consumption via plant uptake, and filtration (Gould et. al., 1994). The exceptionally high surface area found in *sphagnum* has been reported to be an effective ion exchange medium for metal absorption (Vitt and Slack, 1975), and an effective filter of metal hydroxide particulates (Falbo and Weaks, 1990) that may be present in mine effluent.

Dilution of effluent contaminants:

The initial dilution of wastewater discharged from an open channel outfall describes the amount of dilution achieved in an effluent plume due mostly to the combined effects of waste stream momentum (jet-effect), and buoyancy induced mixing (density-effect) (Brusseau et. al., 1996). The rate of initial dilution can also be increased by increases in ambient current velocity at point of release. Often, initial dilution ratios of 100:1 are sought in the design of industrial outfalls to reduce the toxicity of dissolved industrial contaminants to acceptable levels. However such contaminants may also reach the receiving environment by being absorbed onto entrained (suspended) particles in the overlying water, or by being washed from buried sediments during large storm events. The rate of deposition of a waste stream's suspended load is directly related to the entrained particle's size, and is inversely related to the stream's injection velocity (Ward and Eliot, 1995). Deposition rates for suspended contaminants can also be influenced by both physical and chemical resuspension processes, and by site specific geochemical and organic adsorption/desorption equilibria (Schnoor, 1996).

Following initial dilution, river turbulence and ambient currents work to further dilute the waste plume until at some distance below outfall, the effluent reaches an equilibrium with the receiving water. Additional (secondary) dilution of a plume's aqueous constituents is usually achieved relatively quickly in the vertical and longitudinal directions by advection, but occurs only gradually laterally through turbulent diffusion (Gowda, 1980). The rate of effluent mixing by turbulent diffusion may be increased by the presence of stream shear forces caused by vertical and lateral velocity gradients at the bank-water interface. In situations where ambient current velocities are low, other dispersal forces; such as the interaction of microscale turbulence with wind/wave shear, become more important contributors to secondary dilution (Schnoor, 1996).

The area of the receiving river that lies between the point of outfall and the nearest cross section at which the effluent distribution becomes cross-sectionally uniform in concentration has been broadly referred to as a mixing zone by some authors (Gowda, 1984). Traditionally in the water quality practice, a portion of the mixing zone may be delineated as a *zone of limited use*, wherein the concentration of a contaminant may exceed a specified criterion (MOEE, 1994), making the area unsuitable for aquatic life. The prediction of water quality through the prediction of zones of limited use downstream of shallow, river bank, point-source outfalls, has in the past been performed using models that follow a modified stream tube approach (Gowda, 1984). The prediction of sediment quality in riverstrine systems is typically preformed one-dimensionally, with respect to time, by coupling a chemical equilibrium model with mass balance equations (Schnoor, 1996) for a given contaminant at a given distance from outfall. As of yet, a spatial model that combines both sediment and water quality prediction applicable to contaminant dispersion and deposition within a river mixing zone, does not exist.

This thesis examines the effectiveness of modern mining environmental control systems used to reduce potential contaminants, and thereby minimize impact on the aquatic and benthic receiving environments. The focus of this research was to predict both water and sediment quality in a receiving river's mixing zone. Specifically, the project objectives were; 1) to identify and spatially describe aqueous and sediment contaminants of current or potential environmental concern, and 2) to determine if the effluent receiving water body was capable of diluting contaminants to concentrations not significantly different from levels found upstream of discharge.

Methods:

1. Study Area and Site Description

Placer Dome Canada's Musselwhite gold mine is located approximately 150 km north of the town of Pickle Lake, Ontario, Canada (Figure 1).

1.1. Mine Water Management:

At Musselwhite, mine water from underground workings is collected and pumped to the surface where it is used by the mill as process make-up water. A schematic for water management at the mine is presented in Figure 2. Excess mine water is released directly into a tailings pond without prior treatment. The tailings pond, along with receiving both treated mill effluent and untreated mine water, also receives water inputs from camp and mill/plant septic tank sources. Tailings pond decant that is not reclaimed by the mill, is discharged into a polishing pond where additional natural degradation of residual cyanide products, precipitation of contaminants, and sedimentation of suspended solids is promoted. The polishing pond also receives additional water inputs from open pit de-watering activities, waste rock pile runoff, and mill/plant site runoff. Final effluent, released from the polishing pond, is first passed through a wetland area before being discharged into a receiving river.

1.2. Final Discharge Area:

Final, treated effluent at the Musselwhite mine is discharged onto the *shore* of Lake 282 (50deg 36'N, 90deg 21'W), via a diversion channel constructed from the mine's crushed, waste pit rock that runs between the wetland area and the receiving lake (Figure 3). The effluent outfall is designed to be a point source discharge, situated at the most southerly point on the lake's western basin. Lake 282 is formed from a simple widening of the Paseminon

Figure 1. The Musselwhite Mine Location.

Figure 2. Water / Tailings Management System Schematic for Placer Dome's Musselwhite Mine.

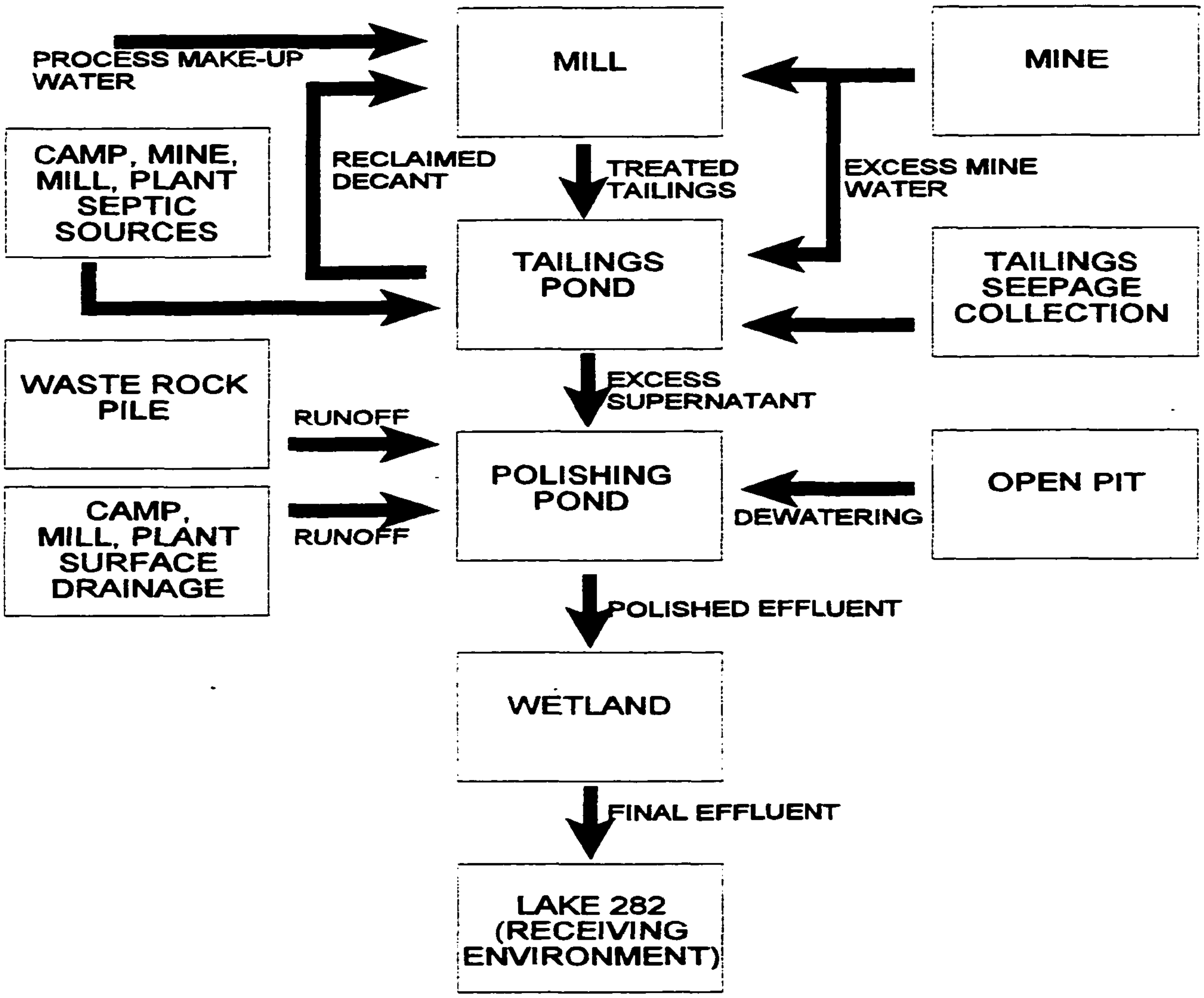
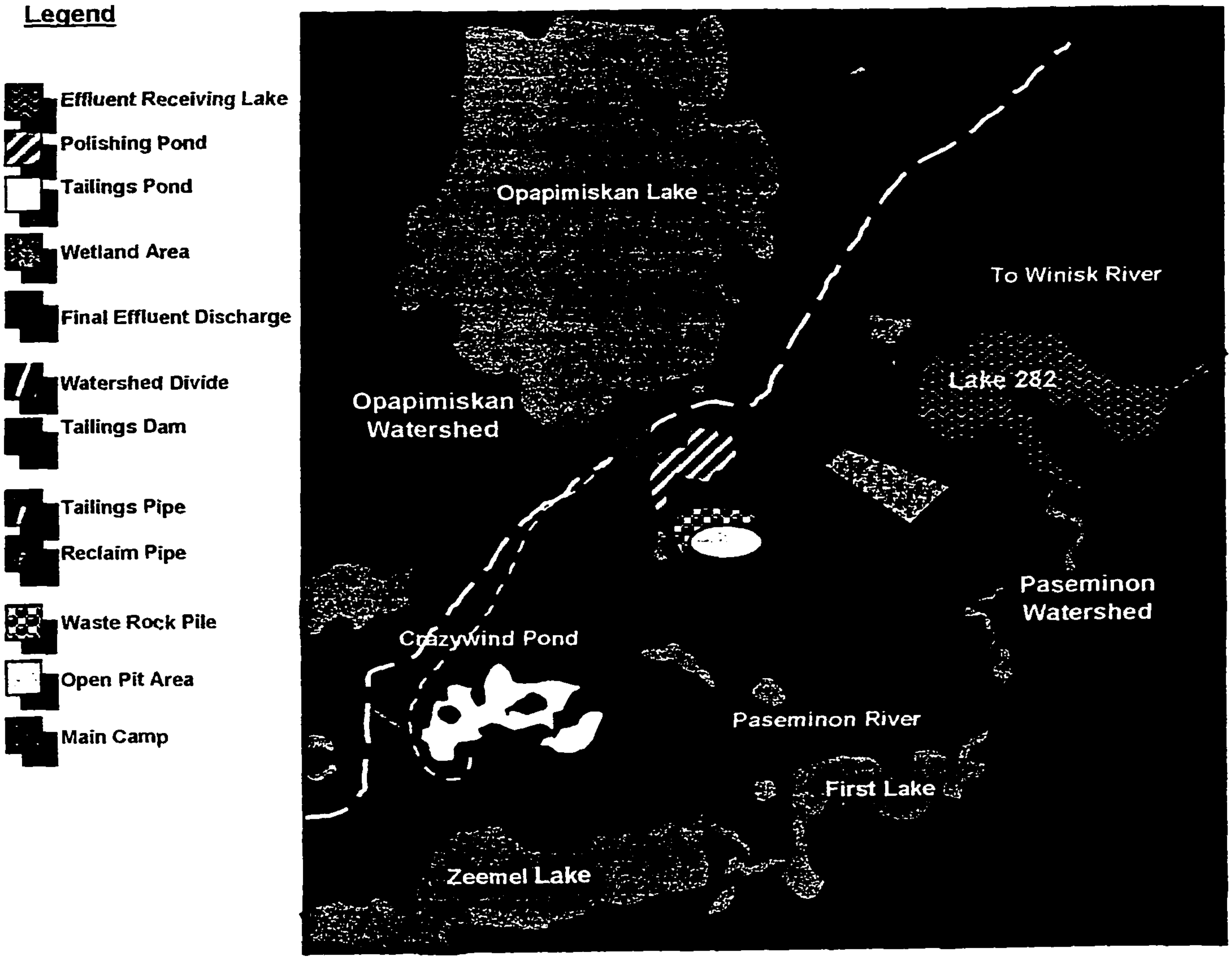


Figure 3. Major Mine Structures, Lakes, and Watersheds in the Musselwhite Project Area



river, a tributary of the Pipestone River, that ultimately drains into Hudson Bay through the Winisk River system (Figure. 3).

1.3. Potential Site Specific Sources and Assessments for AMD;

Potential sources of AMD that may reach Musselwhite's effluent waters include those originating from underground workings, ore stockpiles, tailings, underground waste rock, open pit walls, and open pit waste rock. Between 1988 and 1995, an assessment for the potential of AMD generation at the mine was conducted through the techniques of acid/base accounting, and acid generating kinetics tests performed on samples collected from tailings and waste rock sites. The results of these studies suggested that, at the time of the study, AMD was not being generated by the mine. Tailings were however predicted to be capable of generating acidic drainage after a lag period of 5 years. All AMD generation was predicted by the mine however to be minimal for two reasons; 1) the low mean sulphur content of the main ore deposit (1.85%) and 2) the mine's practice of submerging stored tailings in order to limit sulphide mineral oxidation. Musselwhite's waste rock piles were similarly not expected to be capable of generating AMD, a prediction that was claimed to be consistent with the finding of alkaline paste in samples taken from 5 year old waste rock piles on site.

1.4. Explosives use at Musselwhite;

At the Musselwhite mine, explosives are used during blasting activities both in the underground, and on the surface (open pit), with the dominant form of explosive used being the ammonium nitrate based Magnafrac 3200 Emulsion. Contaminant balance computations performed by Musselwhite have predicted ammonia concentrations as high as 7.16 mg/L to exist within the mine water that are derived exclusively from underground explosive use. In total, approximately 128 kg/month of ammonia, and 149 kg/month of nitrate are released into mine and mill water from underground blasting operations and slurry treatment.

1.5. A Summary of Musselwhite's Enhanced Effluent Control Systems;

1.5.1. Tailings Wash, Cyanide Recycling, and Chemical Treatment:

At the Musselwhite mill, CIP waste tailings are pumped to a two stage counter current decant thickener circuit where up to 40% of the cyanide added during cyanidation is recycled back into the leaching circuit. The washed tailings next move to an INCO SO_2/Air reactor for cyanide destruction and metal precipitation before being deposited into the tailings pond.

1.5.2. Tailings and Polishing Ponds:

Treated tailings at Musselwhite are deposited subaqueously into a shallow tailings pond (Crazywind Pond) where the natural degradation of remaining cyanide, and the sedimentation of both metal hydroxide precipitates as well as other particulates is allowed to occur. Tailings pond supernatant is then reclaimed by the mill and is used further as process water before being released into the mine's polishing pond. The polishing pond acts as the mine's final effluent reservoir where release of waste water into the following wetland and receiving lake can be controlled and monitored. The polishing pond not only accepts treated mill effluent, but also functions as a catchment basin for surface runoff and groundwater draining from both the mill site, and the open pit.

1.5.3. Artificial Wetland:

As a final polishing strategy, effluent released from Musslewhite's polishing pond is first run through a black spruce, ericaceous shrub, *Sphagnum* rich, treed bog (Harris et. al., 1996) before being discharging into the receiving lake.

2. Field Studies;

Field activities for this project were conducted in two parts; the identification and mapping of the effluent plume in the study lake, followed by a water and sediment sampling program conducted within and below the mine's effluent plume. Water and sediment sampling, and effluent tracing studies were performed simultaneously during two of three trips made to the study lake in both 1997 and 1998. The additional trip made during these years was for sample collection only.

2.1. Effluent Plume Studies;

2.1.1. Tracer Dye Injection;

Identification of the effluent plume in Lake 282 involved the injection of the fluorescent tracer dye; Rhodamine WT into the mine's discharge channel at a distance of approximately 120 m upstream of outfall to achieve complete mixing with mine effluent prior to release into the receiving lake. Since vertical mixing is often quickly achieved in turbulent white water streams (Kilpatric and Cobb, 1985), the determination of minimum mixing length required for complete lateral mixing was determined to be a most important calculation for insuring complete mixing of the dye with the discharge water. A minimum mixing length of 18.2 m and 16.4 m was estimated for a 95% optimum lateral mixing in June and August of 1998 respectively using the procedure described in Kilpatric and Cobb (1985) for a center point injection with a mean stream velocity, mean stream width, mean depth, and channel slope described in Table 6.1.1.. The channel slope used in this calculation was estimated from topographical maps of the area.

Formula used to determine minimum distance required for 95% optimum lateral mixing of injected tracer dye by effluent water in a stream (Kilpatric and Cobb, 1985);

$$L_o = K(vB^2/E_z) \qquad (18)$$

(Where L_o is the minimum mixing length in feet, K is the coefficient for optimal mixing selected from Kilpatric and Cobb (1985), v is the mean stream velocity, B is the average stream width in feet, and E_z is the transverse mixing coefficient dependant upon stream slope and stream depth, also selected from Kilpatric and Cobb (1985).)

The calculation for constant rate injection was made using the Kilpatric (1970) formula (19) to achieve an equilibrium concentration of 0.100 mg/L Rhodamine WT in the downstream effluent water during each dye application.

Formula used to determine concentration of dye solution injected into effluent stream to yield a plateau concentration of 0.100 mg/L (Kilpatric, 1970).

$$C = 100Q/5.89x10\text{-}7q \qquad (19)$$

(Where C is the concentration of dye solution injected in mg/L, Q is the discharge of the stream in cubic feet per second, and *q* is the rate of injection of tracer in milliliters per minute.)

For the first plume study, conducted in June 1998, a total of 8.325 L of 10% active dye was applied to the discharge channel continuously at a rate of 7.5 ml/min over a span of 18.5 hours with a peristalsic pump. For the second plume study, conducted in August 1998, 13.686 L of 10% active dye was applied at a continuous rate of 6.2 ml/min with the same pump over the same application period. In both studies a target effluent plateau concentration of 0.100 mg/L was used. Dye application rates were confirmed by pumping into a graduated cylinder over 3 min intervals before application to the discharge channel. Achieved dye concentration in the effluent water was confirmed at point of outfall with a field fluorometer after sufficient dilution of effluent samples (Table 6.1.2.).

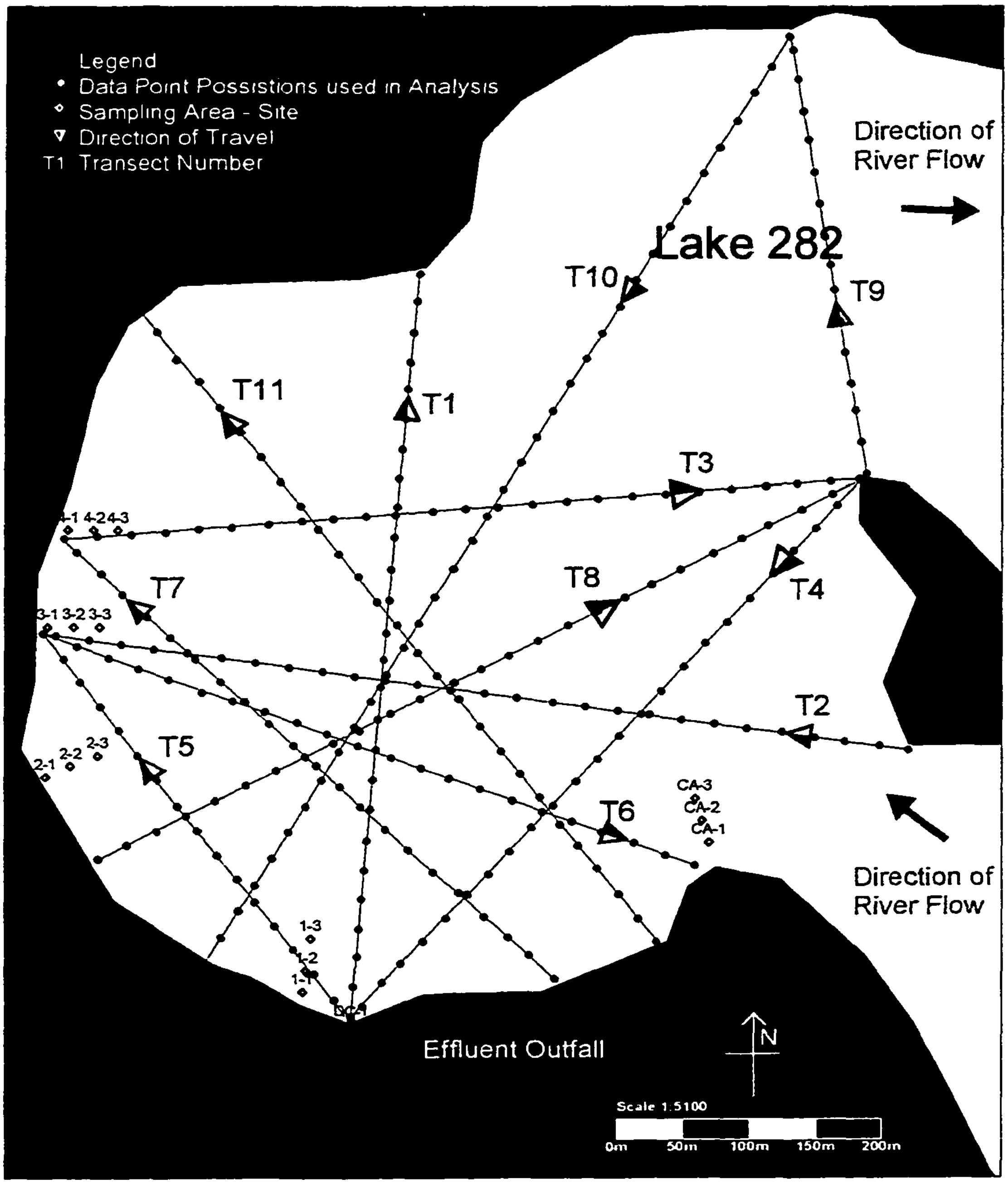
2.1.2. Dye measurements

Dye concentrations in the effluent plume were measured along a set of 11 transects (Figure. 4) with the Turner Designs 10AU Digital Fluorometer after 13 hours of dye application. Transect speed was keep constant and a sampling depth of 1.0 m was used in both studies. Sampling depth was selected based upon the highest concentration of dye in the vertical profile as measured at two points in the lake (Table 6.1.1.). Time and dye concentration data was logged continuously over 5 hours at 3 second intervals while utilizing an online correction for temperature selected specifically for Rhodamine WT (Carter, 1974).

3. Field Procedures:

Prior to sampling, a series of 19 stakes were placed along the western shore of Lake 282 beginning at the point of effluent outfall, and ending at the northern edge of the western basin. These stakes were spaced 40 m apart on the water, at a distance not exceeding 3 m

Figure 4. Transects used to measure 1998 effluent plumes, and sampling areas/sites used on Lake 282.



from shore, and were used in subsequent sampling and plume study transects as shoreline location markers.

3.1 Sampling Areas and Sampling Sites;

Water and sediment sampling on Lake 282 began in July of 1997. In total, six sampling areas were established on the lake (Fig. 4). Areas one through four were located downstream of the effluent outfall. Area five (control, CA) was located approximately 100 m upstream on the river from the effluent diversion channel. The sixth sampling area (DC) was positioned within the discharge channel itself. Each sampling area consisted of three sampling sites marked with floatation markers that were spaced 20 m apart along transects oriented towards the middle of the lake. More accurate information on sampling site position can be found in Table A.2.1. of Appendix 2. The choice of sampling area position was based upon effluent modeling predictions previously made by the mine that were roughly confirmed in the first field season with preliminary plume studies. Sample area and site positions were chosen to lie in the direct path of the effluent-tracer plumes measured during 1997 dye applications (Appendix 3).

In 1998, modifications were made to the dye application procedure. These modifications included the dye application time being lengthened from 7 hours to 18.5 hours to allow for more complete mixing in Lake 282 due to the very low current flow conditions encountered in the lake (Personal Communication, Golder Associates), and the dye injection point moved from 15 m from outfall in 1997, to approximately 120 m from outfall in 1998 to allow for more complete mixing of the dye with effluent water. Attempts to measure current velocity in Lake 282 were made during each trip in 1997 using a 2100-STDX current velocity flow meter purchased from Hoskin Scientific Limited. However, water velocity turned out to be below the instruments detection limit, and measurement attempts were discontinued in 1998.

3.2 Sampling site location;

Sampling site locations are presented in Figure 4. A more precise description of location can be found in Table A.2.1 of Appendix 2.

3.3 Field Procedures;

Water and sediment samples were collected from Lake 282 in July, August and September of 1997, and in June, July, and August of 1998. Water was collected at each site both from the surface, and from the sediment/water interface using a Kemmerer bottle. The top 10 cm of lake sediments were also sampled at each site using an Birge-Ekman dredge. Each water sample taken was further divided into three separate volumes (1 x 1 L, 2 x 250 mL) for the purposes of preservation before being transported back to the Lakehead University Environmental Lab (L.U.E.L.) for analysis. The 1 L water sample was first cooled on ice prior to shipment back to L.U.E.L, and was used to measure; conductivity, pH, nitrate, nitrite, ammonia, total kjedahl nitrogen (TKN), sulphate, and dissolved metals. The first 250 ml sample was preserved with 0.5 ml of HNO_3 (conc) for total metal determinations. The second 250 ml sample was preserved with 9ml of 2N NaOH, cooled to 4 °C, and then insulated from exposure to light for total cyanide, weakly acid dissociable (WAD) cyanide, and thiocyanate determinations. Cyanide samples were the most sensitive to degradation by time, temperature, and light, therefore special care was taken to insure quick preservation, packaging, and storage of these samples (completed within 1 hour of sampling). Collected sediment samples were bagged and frozen within 3 hours of collection. Field sampling quality controls included 1 split sample, 1 field blank, 1 travel blank, and 1 replicate sample taken in each month. Field and travel blanks were taken only during water sampling.

4. Laboratory procedures:

4.1 Water Analysis:

Water Total Kjedahl Nitrogen concentrations were determined spectrophotometrically at 660 nm on the Skalar Autoanalyzer system based upon a modified Berthelot (Greenberg et. al., 1992) reaction. Before analysis, samples were pretreated with strong sulphuric acid, in a stepwise digestion to 400 °C according to Searle (1994), to achieve a three fold preconcentration. Ammonia determination for water was performed using the same technique without prior sulphuric acid digestion and preconcentration.

Combined nitrate and nitrite concentrations were initially determined on the Skalar Autoanlyzer at 540 nm using the cadmium reduction method described in Greenberg et. al, (1992). Nitrite values were estimated independently using the same azo-dye forming reaction but without prior cadmium reduction (Greenberg et. al., 1992). The subtraction of the nitrite concentrations from the combined nitrate and nitrite determination yielded the nitrate values reported to the mine.

Sulphate concentrations in 1997 were measured spectrophotometrically on a Cary 5E UV Spectrophotometer at 420 nm after reacting sulphate ions in solution with barium chloride in an acetic acid medium according to Greenberg et. al. (1992). Ammonia, nitrate, nitrite, and sulphate were analyzed by ion chromatography on a Dionex Model IC in 1998. Sample preparation for these species involved filtering 300 ml of whole water through a Whatman 527 filter papers before presenting to the IC for analysis.

Dissolved metals were analyzed from collected water samples using a Jerrell-Ash Model, inductively coupled ion-spectroscope (ICP) without pretreatment prior to determination.. Total metals analysis was also performed through ICP after a nitric acid digest at 120 °C for 4 hours according to Greenberg et. al. (1992).

In total, three types of cyanide analyses were performed on preserved waters. The measurement of these species involved a modification to procedures described by Wright (1993) due to suspected nitrite interferences encountered during determination. Weakly acid-dissociable cyanide (WAD cyanide) was measured spectrophotometrically on the Skalar Autoanalyzer system at 600 nm according to Meussen et. al. (1988), using a cyanide distillation module installed on the Skalar Autoanalyzer. There was no detectable nitrite interference encountered in the WAD cyanide determination. Total cyanide (Tot-CN) was measured using the same procedure as WAD cyanide, with the addition of an online UV digest also according to Meussen et. al (1989). A response ratio of 5:1 was determined to be likely attributed to nitrite interference during this test and total cyanide values were corrected only after both nitrite and cyanide determinations. Thiocyanate was measured using the same procedure for WAD cyanide after pretreating 10 ml samples with 50 micro liters of 3.7% formaldehyde for 24 hours prior to determination (Meussen et. al., 1988).

4.2 Sediment Analysis;

Collected sediment samples were thawed in a warm water bath and oven dried on paper plates at 40 °C for a minimum of 48 hours, or until dry, according to the pretreatment procedure for cyanide defined in Wright (1993). Samples were then ground with a mortar and pestle, sieved though a 2 mm diameter soil sieve before being divided into suitable amounts for analysis.

Sediment total metal values were determined through the following procedure. Sample preparation involved ashing 0.5 g of oven dried sediment at 600 degrees C for 16 hours, followed by heating to 80°C, for 4 hours, in 2 ml of digestion acid consisting of a 1:3 mix of HNO_3 to HCl. Loss upon ignition was also determined during this procedure by weighing samples prior to, and after ashing Following digestion, samples were cooled and diluted with distilled deionized water before being presented to ICP for analysis.

The procedure for the determination of extractable metals in sediment was performed according to the American Soil and Plant Analysis Council (1992) for hard metals extraction in acidic soils ($pH < 7$). In this procedure, 2.5 g of dried sediment was shaken on a rotational shaker for 30 min in the presence of 50 ml of 0.1 N HCl. Extracting solution was then filtered through Whatman #1 filters before being presented to ICP for analysis.

Water extractions were performed for nitrate and sulphate on sediments, in which 5 g of dry sediment was shaken for 4 hours on a rotational shaker in the presence of 50 ml of distilled deionized water before being filtered through Whatman #1 filters according to. Extracts were then taken to IC for analysis.

Extractions for cyanide determination in dried sediment were performed using a modification to the procedure defined in Wright (1993). Cyanide was extracted from 2.5 g sediment in 50 ml of 1% NaOH for 16 hours using a rotational shaker to achieve a liquid to solid ratio of 20:1. Extraction ratios between 4:1 and 20:1 have been reported to have no influence on extraction yield when an extraction time of 16 hours is used Wright (1993). Extracts were filtered through Whatman #1 type filter papers, and then diluted 10 fold with DDW before being passed through a strong anion exchange cartridge in order to further remove organic acids from solution. Samples were then diluted an additional five fold with sample matrix (0.1% NaOH) before being presented to the Skalar system for analysis. The analysis for cyanide parameters on these soil extracts were then performed according to the same procedures described in the water analysis.

5. Data Analysis:

5.1 Plume Data Interpretation;

After data collection, logged dye concentrations were re-expressed as a percent of effluent concentration through a simple transformation. The position of each data point along a given transect was estimated by first calculating the individual transect velocity, and then multiplying by the logged elapsed time. Percent effluent transect data was then transferred to Golden Software's SURFER for Windows 95 for further analysis. An interval of 20 m between data points on each transect was chosen to generate the two-dimensional contour plots produced through scattered data interpolation. Of the methods available, *Kriging* was chosen for its flexibility in gridding irregularly spaced data (Cressie, 1991) such as that collected along straight line transects. For the purposes of this study, two dimensional plots of both plumes (Figures 6.1.1 & 6.1.2), as well as one composite plume were generated (Figure 6.1.3). The composite plume consisted of an average of the percent effluent values taken from both 1998 plumes.

5.2 Water and Sediment Data Sets & The Common Variable Data Set;

A total of six water and sediment chemistry data sets were generated in this study, consisting of three data sets generated from each year of sample collection. From each of these six data sets, a reduced sub-set of parameters, referred the common variable data set, was constructed which included; Total Aluminum, Total Copper, Total Zinc, Total Manganese, Total Iron, Dissolved Sulphate, Nitrate, and pH measured values. This data set was representative of variables that were measured at all three lake depths.

5.3 Data Set Analysis;

The following statistical analyses and transformations were performed using SPSS 9.0 for Windows 95;

1) A series of two way, between subjects analysis of variance (ANOVA) was conducted to test for the main effects of sampling time and sampling location on each parameter measured. A post hoc ANOVA, least significant difference test (LSD) was also performed on each parameter to help further assess differences in group means as defined by sampling area.

2) A graphical analysis of the common variable data set was performed using boxplots to compare mean parameter concentrations between sampling areas over sampling time. From this analysis, it was determined that total manganese was a poor candidate for inclusion into the discriminant analysis that followed, and this parameter was removed from the common variable data set.

3) A correction for the influence of sampling time was performed according to Lee and Stewart (1981) on those remaining common variable data set parameters identified through ANOVA as being significantly influenced by time. The correction consisted of a transformation in which each case was reexpressed as a percent of the mean value for all cases in a given month.

4) Discriminant analysis was then used to further test the common variable data set for the existence of groups as defined by sampling area after time correction. Wilks Lambda was

used as the separating statistic among group centroids for this test. From the discriminant functions generated, the first function consistently summarized a significant proportion of the variation among groups. However, the signs of the discriminant function coefficients were not consistent in all analyses. In order to allow discriminant function score data to be used in the next phase of the analysis, a sign transformation was applied to three of the six sets of discriminant function scores to insure consistency between the first function axis and contributing discriminating variables.

6) The 1997 and 1998 means for monthly, sign transformed, discriminant function score at each site were then calculated, and plotted against the natural log of the 1998 mean percent effluent value at each site. A simple linear regression analysis was then used to relate the mean percent effluent to discriminant function score.

7) Plots of predicted discriminant function score for each of the six data sets were then produced in SURFER after transforming the mean percent effluent data through the linear regression equations generated.

8) Compliance with provincial effluent limits, provincial water quality objectives (PWQO's), and provincial sediment quality guidelines (PSQG's) for select parameters at select sites was also examined independently in this study using monthly and yearly mean values (Appendix 1) .

Results;

6.1 Measurements and calculations related to 1998 effluent plume studies.

The calculation of minimum discharge channel mixing length required for 95% lateral mixing in a stream by an injected tracer dye was made using the formula described in Kilpatric and Cob (1985) with measurements included in Table 6.1.1..

Table 6.1.1. Calculations for minimum discharge channel mixing length required for 95% lateral mixing of injected tracer dye.

Plume Study	Mean (n=3) Stream Width (m)	Stream Slope (m/m)	Mean (n=3) Stream Depth (m)	Stream Velocity (m/sec)	Transverse Mixing Coefficient*	Coefficient for 95% Mixing from a Center Injection*	Minimum Required Mixing Length (m)
1. June	1.3	0.0133	0.30	1.0	0.1	0.100	18.2
2. Aug.	1.3	0.0133	0.25	0.9	0.1	0.100	16.4

(*From Kilpatric and Cobb, 1985)

Possibly due to inaccuracies in the estimation of channel velocity and other stream mixing characteristics, achieved dye concentration in the effluent at point of outfall differed from that predicted by the Kilpatric (1970) formula's estimation of plateau concentration. In both years, achieved dye concentrations turned out to be higher than that expected (Table 6.1.2.), yet resulted in effluent-tracer plumes in the receiving lake that were within the detection range required for flow-through fluorometric measurement. Lake vertical profile measurements of Rhodamine WT concentration, taken just outside of outfall, showed the highest concentration of dye to exist at a depth of 1 m during both plume studies (Table 6.1.2.).

Table 6.1.2. Achieved effluent concentrations for RhodamineWT in both discharge channel and vertical lake profile, and Pàseminon river flow conditions reported at outlet during 1998 effluent plume studies.

		Discharge Channel Dye Concentration Measurements		Vertical Profile Dye Concentration Measurements in Lake 282 (mg/L)				
Plume Study /Date	River Flow Rate (m^3 /min)	Desired Effluent Concentration (mg/L)	Achieved Effluent Concentration (mg/L)	Sampling Distance from Outfall (m)	Depth			
					0 m	0.5 m	1.0 m	1.5 m
1/June 1998	50500	0.100	0.328	60	0.00141	0.00220	0.00442	0.00340
2/Aug 1998	28805	0.100	0.126	60	0.00151	0.00170	0.00182	0.00100

6.1 Effluent Plume Plots

Plots of *kringing* predicted mine effluent concentration in the western basin of Lake 282 during July and August of 1998 (Figures 6.1.1. & 6.1.2) suggested a strong influence of season on the dilution of effluent at the 1 m depth. In both plots, an area of highest effluent concentration, possibly a region of initial effluent dilution, was predicted to extend from Sample Area 1 upstream to just prior to the Control Area.

The June mine water effluent plume (Fig. 6.1.1.), possessing both the highest initial effluent dilution ratio in the vicinity of the outfall and the greatest secondary dilution, appeared to have considerable lateral dispersion, and is likely a product of the higher flow conditions that exist in the Paseminon River during that time of year. Mine effluent likely only reaches the western shore of Lake 282 in very low concentrations (0.05-0.10%) during the mine's early discharge season (May to June), and is quickly reduced to what may approach equilibrium concentrations (0.05-0.50%) at a distance of less than 100 m north from the outfall. A maximum effluent concentration of 4.00% was measured within 10 m of the outfall during this application. Contours for slightly lower concentrations (0.50-1.00%) were limited to a narrow area along the southern edge of the western basin, and are believed to describe the extent of initial dilution.

The August mine water effluent plume (Fig. 6.1.2.), having been performed later in the season, best shows the effect of lower river flow conditions on effluent dispersion at the 1 m depth. Secondary dilution in August appeared to require a much longer reach to achieve than what was required at the beginning of the season, and possible near equilibrium concentrations (<0.10%) of mine effluent were not observed until the plume had dispersed longitudinally approximately 600 m from outfall. Along the western shore of the lake, effluent concentrations typically ranged from between 0.70-0.90%, and maximum concentration of 7.5% was measured within 10 m of outfall. The extent of initial dilution, thought to be best delineated by the 1.00% effluent contour, was predicted to be approximately double that predicted for June.

The 1998 predicted mean mine water effluent plume presented in Figure. 6.1.3. was further used to produce the models of contaminant deposition in Lake 282.

Figure 6.1.1. Predicted Mine Water Effluent Plume, June 1998, Depth 1m.

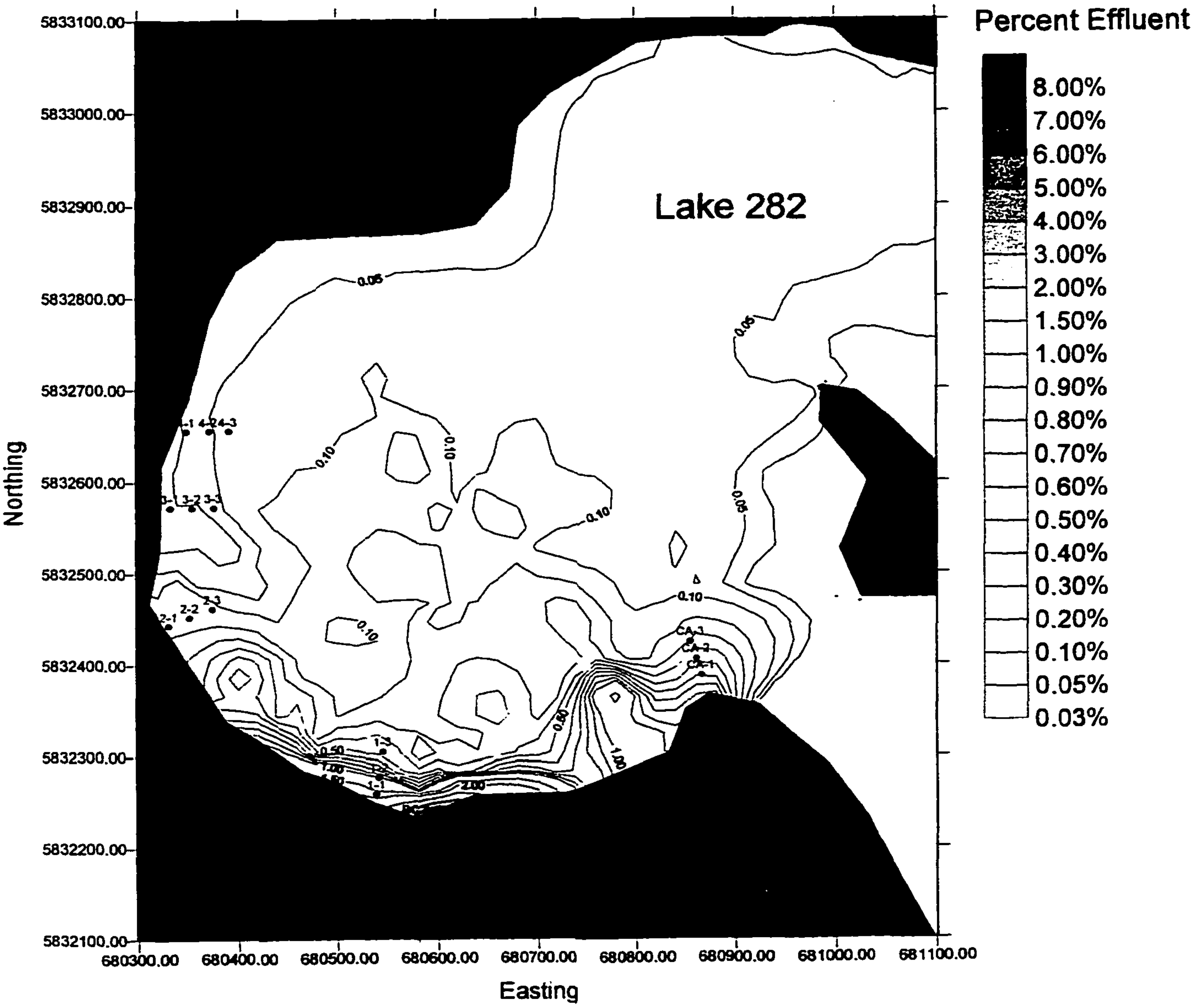


Figure 6.1.2. Predicted Mine Effluent Plume, August 1998, Depth 1m.

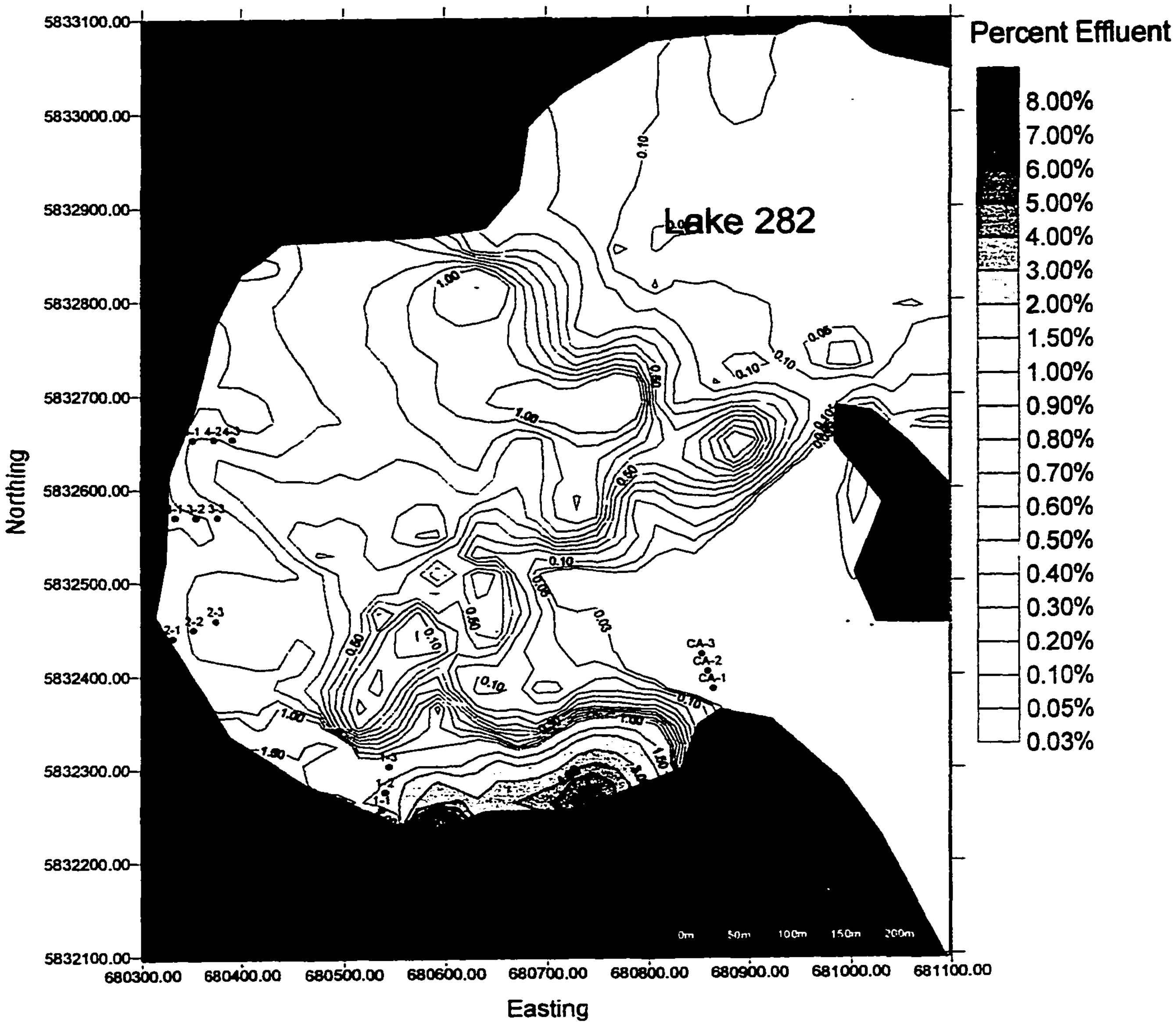
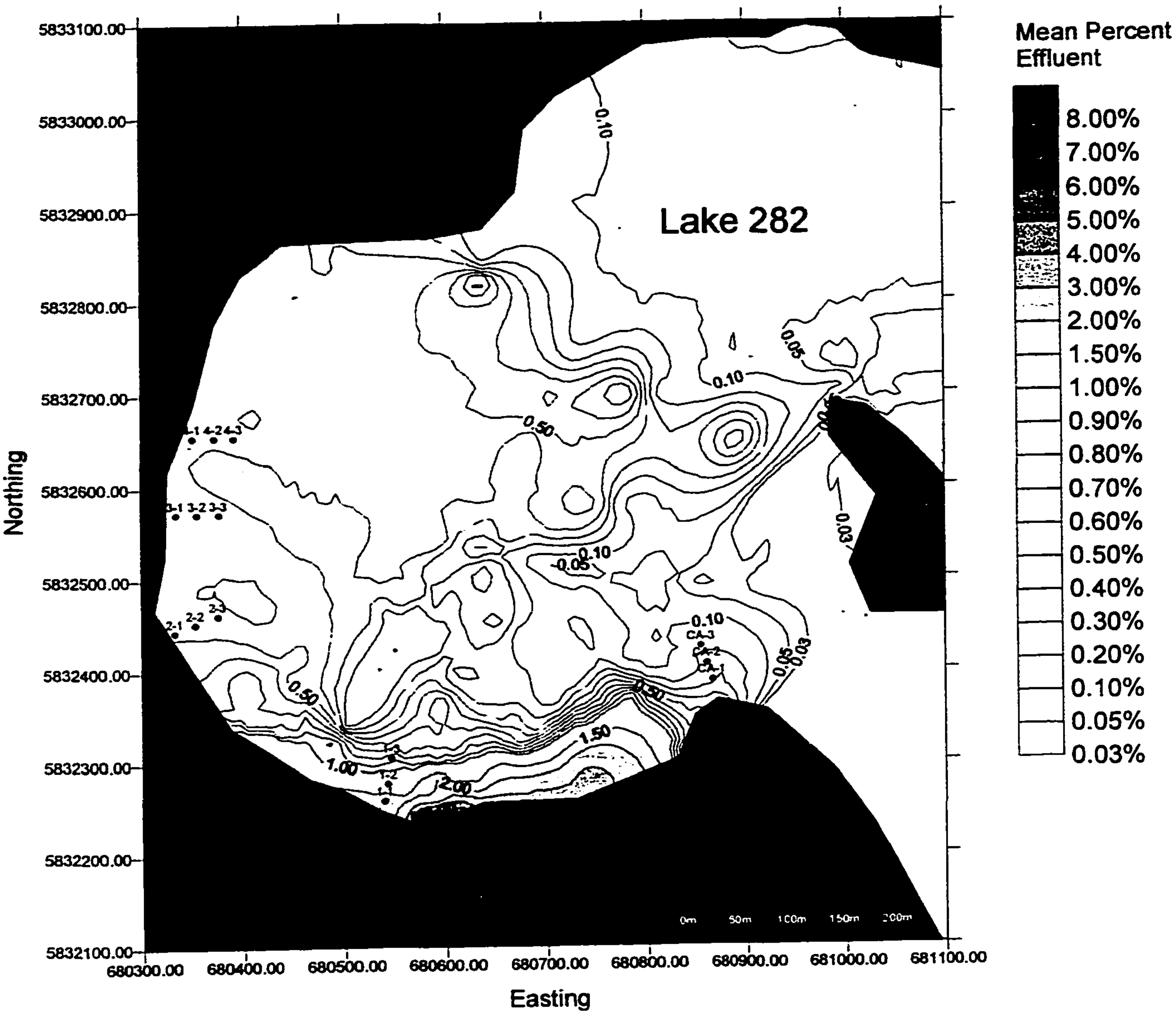


Figure 6.1.3. Predicted, 1998 Mean Effluent Plume, Depth 1m.



6.2 ANOVA and LSD results:

To test for the influence of season and location on individual lake parameters, a series of two-way ANOVA's were performed on each water and sediment quality data set (Tables 6.3.1-.6.3.6.). Differences between group means defined by sampling area were also assessed through Least Significant Difference tests. Those parameters with significance effects listed as Below Detection Limit (BDL) were not included in the analysis due to a majority of missing values or a majority of cases falling below detection limits, resulting in too few cases available for testing.

Table 6.2.1 Mean values for 1997 surface water data set parameters. All values in (mg/L) except conductivity (expressed in microohms/cm^2), and paste pH (expressed in pH units). ANOVA results are represented by; *, denotes a significant effect ($p<0.05$, df=4/40) of sample area on parameter; **, denotes a significant effect ($p<0.05$, df=2/48) of sample time (month) on parameter. Population means with the same letter suffix are not significantly different ($p<0.05$) as determined by a LSD test. BDL = Mean values below detection limit.

Parameter	Mean Conc. Area 1	Stnd. Dev. in Mean Area 1	Mean Conc. Area 2	Stnd. Dev. in Mean Area 2	Mean Conc. Area 3	Stnd. Dev. in Mean Area 3	Mean Conc. Area 4	Stnd. Dev. in Mean Area 4	Mean Conc. Control	Stnd. Dev. in Mean Control
Total Aluminum **	0.106 a	0.034	0.107 a	0.044	0.125 a	0.048	0.095 a	0.032	0.089 a	0.028
Total Copper **	0.005 a	0.003	0.004 a	0.001	0.003 a	0.001	0.004 a	0.003	0.004 a	0.002
Total Iron	0.122 a	0.043	0.133 a	0.043	0.182 a	0.118	0.119 a	0.031	0.201 a	0.202
Total Manganese **	0.011 a	0.003	0.012 a	0.002	0.014 a	0.008	0.011 a	0.002	0.021 a	0.028
Total Nickel	BDL		BDL		BDL		BDL		BDL	
.Total Ziinc *, **	0.008 a	0.003	0.003 b	0.001	0.003 b	0.002	0.003 b	0.002	0.003 b	0.002
TKN	0.266 b	0.052	0.298 ab	0.088	0.332 ab	0.152	0.384 a	0.199	0.248 b	0.056
Nitrate	0.219 a	0.218	0.169 a	0.174	0.129 a	0.142	0.145 a	0.181	0.175 a	0.231
Nitrite	BDL		BDL		BDL		BDL		BDL	
Ammonia	BDL		BDL		BDL		BDL		BDL	
Sulphate	9.751 a	4.294	10.126 a	3.634	11.149 a	4.978	11.512 a	4.189	12.789 a	8.418
pH *	7.34 b	0.11	7.45 c	0.08	7.36 bc	0.14	7.35 bc	0.13	7.47 a	0.08
Conductivity **	77.022 a	23.462	75.244 a	21.879	86.833 a	33.799	76.267 a	19.149	75.700 a	26.062
Dissolved Aluminum	BDL		BDL		BDL		BDL		BDL	
Dissolved Copper	BDL		BDL		BDL		BDL		BDL	
Dissolved Iron **	0.043 a	0.011	0.045 a	0.011	0.043 a	0.014	0.043 a	0.014	0.044 a	0.011
Dissolved Manganese **	0.002 a	0.001	0.002 a	0.001	0.001 a	0.001	0.002 a	0.001	0.001 a	0.001
Dissolved Nickel	BDL		BDL		BDL		BDL		BDL	
Dissolved Zinc	BDL		BDL		BDL		BDL		BDL	
Total Cyanide	BDL		BDL		BDL		BDL		BDL	
WAD Cyanide	BDL		BDL		BDL		BDL		BDL	
Thiocyanate	BDL		BDL		BDL		BDL		BDL	

Table 6.2.2 Mean values for 1998 surface water data set parameters. All values in (mg/L) except conductivity (expressed in microohms/cm^2), and paste pH (expressed in pH units). ANOVA results are represented by; *, denotes a significant effect ($p<0.05$, df=4/40) of sample area on parameter; **, denotes a significant effect ($p<0.05$, df=2/48) of sample time (month) on parameter. Population means with the same letter suffix are not significantly different ($p<0.05$) as determined by a LSD test. BDL = Mean values Below Detection Limit.

Parameter	Mean Conc. Area 1	Stnd. Dev. in Mean Area 1	Mean Conc. Area 2	Stnd. Dev. in Mean Area 2	Mean Conc. Area 3	Stnd. Dev. in Mean Area 3	Mean Conc. Area 4	Stnd. Dev. in Mean Area 4	Mean Conc. Control	Stnd. Dev. in Mean Control
Total Aluminum	0.071 a	0.005	0.065 a	0.021	0.048 b	0.014	0.058 ab	0.025	0.066 a	0.018
Total Copper	0.007 a	0.006	0.006 a	0.003	0.005 a	0.002	0.004 a	0.001	0.005 a	0.001
Total Iron	0.093 a	0.017	0.121 a	0.057	0.095 a	0.016	0.120 a	0.068	0.108 a	0.021
Total Manganese	0.023 a	0.025	0.017 a	0.007	0.014 a	0.005	0.017 a	0.009	0.015 a	0.004
Total Nickel	BDL		BDL		BDL		BDL		BDL	
Total Ziinc *	0.009 a	0.006	0.004 b	0.002	0.003 b	0.002	0.003 b	0.001	0.004b	0.001
TKN **	0.494 a	0.196	0.539 a	0.233 a	0.634 a	0.416	0.812 a	0.320	0.750 a	0.506
Nitrate **	0.276 a	0.338	0.254 a	0.324	0.234 a	0.299	0.416 a	0.564	0.320 a	0.424
Nitrite	BDL		BDL		BDL		BDL		BDL	
Ammonia	BDL		BDL		BDL		BDL		BDL	
Sulphate **	15.943 a	21.318	14.911 a	20.508	14.155 a	19.246	23.218 a	32.871	18.471 a	25.651
pH **	7.14 a	0.36	7.35 a	0.28	7.39 a	0.35	7.39 a	0.27	7.45 a	0.35
Conductivity **	118.355 a	73.561	113.222 a	70.101	119.744 a	75.987	147.578 a	118.881	123.844 a	84.659
Dissolved Aluminum	BDL		BDL		BDL		BDL		BDL	
Dissolved Copper	BDL		BDL		BDL		BDL		BDL	
Dissolved Iron	0.046 a	0.09	0.042 ab	0.014	0.030 b	0.018	0.040 ab	0.014	0.046 a	0.012
Dissolved Manganese **	0.002 ab	0.001	0.002 b	0.001	0.002 ab	0.001	0.003 a	0.001	0.002 ab	0.001
Dissolved Nickel	BDL		BDL		BDL		BDL		BDL	
Dissolved Zinc	BDL		BDL		BDL		BDL		BDL	
Total Cyanide	BDL		BDL		BDL		BDL		BDL	
WAD Cyanide	BDL		BDL		BDL		BDL		BDL	
Thiocyanate	BDL		BDL		BDL		BDL		BDL	

Table 6.2.3 Mean values for 1997 sediment/water interface data set parameters. All values in (mg/L) except conductivity (expressed in microohms/cm^2), and paste pH (expressed in pH units). ANOVA results are represented by; *, denotes a significant effect ($p<0.05$, df=4/40) of sample area on parameter; **, denotes a significant effect ($p<0.05$, df=2/48) of sample time (month) on parameter. Population means with the same letter suffix are not significantly different ($p<0.05$) as determined by a LSD test. BDL = Mean values Below Detection Limit.

Parameter	Mean Conc. Area 1	Stnd. Dev. in Mean Area 1	Mean Conc. Area 2	Stnd. Dev. in Mean Area 2	Mean Conc. Area 3	Stnd. Dev. in Mean Area 3	Mean Conc. Area 4	Stnd. Dev. in Mean Area 4	Mean Conc. Control	Stnd. Dev. in Mean Control
Total Aluminum *	0.199 a	0.141	0.128 bc	0.041	0.181 ac	0.033	0.108 b	0.036	0.123 bc	0.036
Total Copper *, **	0.006 a	0.003	0.005 ab	0.002	0.004 ab	0.002	0.003 b	0.002	0.004 ab	0.002
Total Iron	0.149 ab	0.031	0.193 ab	0.106	0.266 a	0.154	0.189 b	0.149	0.152 b	0.044
Total Manganese	0.013 a	0.002	0.018 a	0.009	0.021 a	0.009	0.015 a	0.012	0.018 a	0.009
Total Nickel	BDL		BDL		BDL		BDL		BDL	
Total Zinc *, **	0.005 a	0.002	0.004 ab	0.001	0.003 b	0.001	0.003 b	0.001	0.003 b	0.001
TKN *, **	0.271 bc	0.059	0.298 ab	0.107	0.401 ac	0.167	0.417 a	0.237	0.185 b	0.123
Nitrate **	0.306 a	0.261	0.157 a	0.168	0.129 a	0.147	0.137 a	0.163	0.213 a	0.258
Nitrite	BDL		BDL		BDL		BDL		BDL	
Ammonia	BDL		BDL		BDL		BDL		BDL	
Sulphate **	12.702 a	8.211	10.336 a	3.662	11.497 a	4.723	10.197 a	2.281	15.226 a	6.537
pH *, **	7.35 b	0.11	7.43 ab	0.11	7.33 b	0.17	7.34 b	0.13	7.51 a	0.08
Conductivity **	70.933 ab	19.233	69.089 b	17.129	81.111 ab	34.886	71.822 ab	18.204	97.289 a	45.415
Dissolved Aluminum	BDL		BDL		BDL		BDL		BDL	
Dissolved Copper	BDL		BDL		BDL		BDL		BDL	
Dissolved Iron ii)	0.045 a	0.019	0.045 a	0.014	0.045 a	0.016	0.046 a	0.017	0.032 a	0.022
Dissolved Manganese	.0014 a	.0011	.0012 a	.0004	.0011 a	.0004	.0015 a	.0010	.0010 a	.0008
Dissolved Nickel	BDL		BDL		BDL		BDL		BDL	
Dissolved Zinc	BDL		BDL		BDL		BDL		BDL	
Total Cyanide	BDL		BDL		BDL		BDL		BDL	
WAD Cyanide	BDL		BDL		BDL		BDL		BDL	
Thiocyanate	BDL		BDL		BDL		BDL		BDL	

Table 6.2.4 Mean values for 1998 sediment/water interface data set parameters. All values in (mg/L) except conductivity (expressed in microohms/cm^2), and paste pH (expressed in pH units). ANOVA results are represented by; *, denotes a significant effect ($p<0.05$, df=4/40) of sample area on parameter; **, denotes a significant effect ($p<0.05$, df=2/48) of sample time (month) on parameter. Population means with the same letter suffix are not significantly different ($p<0.05$) as determined by a LSD test. BDL = Mean values Below Detection Limit.

Parameter	Mean Conc. Area 1	Stnd. Dev. in Mean Area 1	Mean Conc. Area 2	Stnd. Dev. in Mean Area 2	Mean Conc. Area 3	Stnd. Dev. in Mean Area 3	Mean Conc. Area 4	Stnd. Dev. in Mean Area 4	Mean Conc. Control	Stnd. Dev. in Mean Control
Total Aluminum *	0.343 a	0.333	0.089 b	0.049	0.118 b	0.069	0.153 b	0.078	0.105 b	0.033
Total Copper **	0.007 a	0.003	0.005 bc	0.003	0.005 bc	0.002	0.005 bc	0.002	0.007 ac	0.002
Total Iron *	0.194 abc	0.111	0.327 a	0.261	0.163 bc	0.083	0.293 ab	0.166	0.141 c	0.031
Total Manganese	0.027 a	0.025	0.053 a	0.072	0.015 a	0.006	0.026 a	0.012	0.018 a	0.008
Total Nickel	BDL		BDL		BDL		BDL		BDL	
Total Ziinc *, **	0.007 a	0.002	0.005 b	0.002	0.004 b	0.002	0.003 b	0.001	0.005 b	0.001
TKN **	0.489 a	0.231	0.574 a	0.435	0.831 a	0.427	0.651 a	0.396	0.804 a	0.565
Nitrate **	0.355 a	0.491	0.308 a	0.338	0.219 a	0.282	0.309 a	0.333	0.288 a	0.338
Nitrite	BDL		BDL		BDL		BDL		BDL	
Ammonia	BDL		BDL		BDL		BDL		BDL	
Sulphate **	20.199 a	30.382	14.279 a	19.391	26.860 a	50.133	17.372 a	25.003	21.736 a	30.313
pH **	7.12 a	0.30	7.25 ab	0.37	7.34 ab	0.33	7.41 b	0.21	7.45 b	0.23
Conductivity **	138.200 a	109.801	109.500 a	63.591	144.366 a	142.435	130.766 a	92.918	147.133 a	121.882
Dissolved Aluminum	BDL		BDL		BDL		BDL		BDL	
Dissolved Copper	BDL		BDL		BDL		BDL		BDL	
Dissolved Iron	0.053 a	0.018	0.036 a	0.016	0.039 a	0.024	0.048 a	0.001	0.041 a	0.019
Dissolved Manganese **	0.002 a	0.001	0.002 a	0.001	0.002 a	0.001	0.002 a	0.001	0.002 a	0.001
Dissolved Nickel	BDL		BDL		BDL		BDL		BDL	
Dissolved Zinc	BDL		BDL		BDL		BDL		BDL	
Total Cyanide	BDL		BDL		BDL		BDL		BDL	
WAD Cyanide	BDL		BDL		BDL		BDL		BDL	
Thiocyanate	BDL		BDL		BDL		BDL		BDL	

Table 6.2.5 Mean values for 1997 sediment data set parameters. All values in (mg/kg) dry weight sediment except LOI (expressed as % dry weight) and paste pH (expressed in pH units). ANOVA results are represented by; *, denotes a significant effect ($p<0.05$, df=4/40) of sample area on parameter; **, denotes a significant effect ($p<0.05$, df=2/48) of sample time (month) on parameter. Population means with the same letter suffix are not significantly different ($p<0.05$) as determined by a LSD test.

Parameter	Mean Conc. Area 1	Stnd. Dev. in Mean Area 1	Mean Conc. Area 2	Stnd. Dev. in Mean Area 2	Mean Conc. Area 3	Stnd. Dev. in Mean Area 3	Mean Conc. Area 4	Stnd. Dev. in Mean Area 4	Mean Conc. Control	Stnd. Dev. in Mean Control
Extractable Aluminum **	885.269 ab	233.239	1013.844 a	366.474	656.477 b	146.177	870.349 ab	315.071	869.331 ab	272.084
Extractable Copper *	1.956 bc	.8323	2.807 a	.914	1.468 b	.522	2.048 bc	.400	2.327 ac	.574
Extractable Iron *	175.101 a	100.621	345.279 b	188.419	305.515 b	96.448	284.897 ab	100.475	334.672 b	77.689
Extractable Nickel *, **	2.237 a	.802	1.934 ac	.629	1.261 b	.162	1.568 bc	.469	1.535 bc	.606
Extractable Manganese *, **	423.791 a	173.011	290.263 ab	183.493	203.173 b	79.039	296.621 ab	141.884	416.222 a	203.909
Extractable Zinc	6.006 a	1.164	7.542 a	2.875	5.953 a	1.332	6.655 a	2.421	6.378 a	1.851
Total Aluminum *	11879.979 a	4431.524	5914.383 b	1725.647	4871.152 b	2027.091	5426.929 b	1993.454	6302.231 b	1458.011
Total Copper *	13.631 a	3.724	11.044 ac	1.575	10.201 bc	2.802	9.921 bc	2.851	8.361 b	2.257
Total Iron *	22217.538 a	5849.454	11151.261 b	3795.932	8868.359 b	3078.766	10779.129 b	4525.593	15967.757 c	3228.752
Total Nickel *	23.869 a	6.782	14.757 bc	5.14	11.232 b	3.908	13.788bc	5.479	18.428 c	4.124
Total Manganese *	2430.043 a	882.110	593.494 b	221.815	409.886 b	134.683	571.627 b	246.819	2111.084 a	531.409
Total Zinc *	60.3611 a	17.877	35.568 b	11.961	28.557 b	10.371	31.206 b	13.665	36.058 b	10.533
Nitrate *	5.185 b	1.924	26.413 a	30.167	10.204 b	7.138	6.511 b	4.1486	4.609 b	2.654
Ammonia *, **	19.008 c	8.246	39.941 bc	33.818	80.277 a	50.137	51.746 ab	45.165	14.9019 c	7.6022
Loss Upon Ignition *	15.333 c	6.245	37.333 bc	11.916	41.155 a	8.545	30.444 ab	10.944	11.244 c	5.191
Paste pH *	6.83 a	.25	6.37 b	.07	6.49 bc	.09	6.53 c	.14	6.58 c	.07
Sulphate *	120.513 b	19.349	398.462 a	282.107	178.293 b	67.587	137.943 b	81.024	64.477 b	18.281
WAD Cyanide	BDL		BDL		BDL		BDL		BDL	
Total Cyanide	BDL		BDL		BDL		BDL		BDL	
Thiocyanate	BDL		BDL		BDL		BDL		BDL	

Table 6.2.6 Mean values for 1998 sediment data set parameters. All values in (mg/kg) dry weight sediment except LOI (expressed as % dry weight) and paste pH (expressed in pH units). ANOVA results are represented by; *, denotes a significant effect ($p<0.05$, df=4/40) of sample area on parameter; **, denotes a significant effect ($p<0.05$, df=2/48) of sample time (month) on parameter. Population means with the same letter suffix are not significantly different ($p<0.05$) as determined by a LSD test.

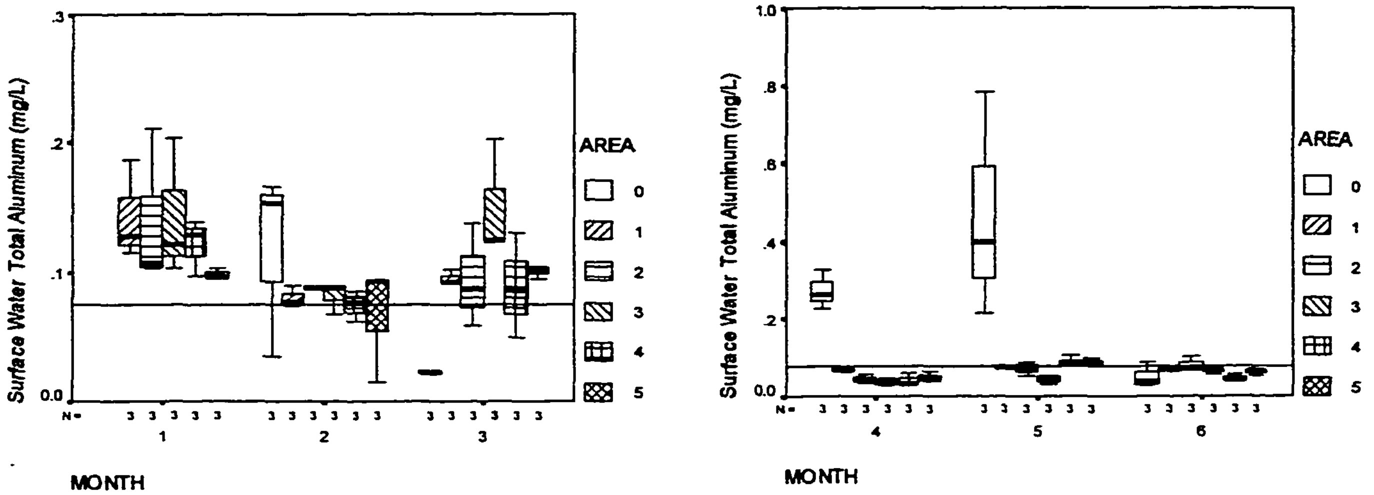
Parameter	Mean Conc. Area 1	Stnd. Dev. in Mean Area 1	Mean Conc. Area 2	Stnd. Dev. in Mean Area 2	Mean Conc. Area 3	Stnd. Dev. in Mean Area 3	Mean Conc. Area 4	Stnd. Dev. in Mean Area 4	Mean Conc. Control	Stnd. Dev. in Mean Control
Extractable Aluminum	727.444 a	215.393	915.889 b	228.732	760.222 ab	138.059	744.000 ab	160.349	763.556 ab	224.462
Extractable Copper *	2.441 a	0.337	2.569 a	0.503	1.830 b	0.401	1.928 b	0.453	2.400 a	0.533
Extractable Iron	243.427 ac	102.544	301.889 c	58.158	328.333 b	46.141	253.556 ac	69.816	315.845 b	64.675
Extractable Nickel *	5.067 a	1.099	4.810 a	0.885	4.801 a	0.756	4.514 a	0.847	3.624b	0.890
Extractable Manganese *	505.556 a	226.997	317.556 bc	87.105	246.444c	48.013	347.778 bc	126.656	399.826 ab	114.625
Extractable Zinc	5.157 a	1.208	7.013 b	2.956	6.798 ab	1.063	5.921 ab	1.546	5.650 ab	1.489
Total Aluminum *	26882.222 a	4306.082	19413.000 b	2999.478	18118.423 bc	4790.275	15722.222 c	3301.431	17365.556 bc	3227.178
Total Copper *	17.389 a	2.889	14.035 b	2.277	13.127 bc	1.955	12.744bc	3.098	11.577c	2.453
Total Iron *	33644.444 ab	8669.214	28022.222 bc	8273.569	22516.540 c	5979.803	22533.333 c	7105.632	36888.889 a	6778.356
Total Nickel *	108.689 a	23.117	77.433 ab	13.647	78.307 b	22.034	68.633 b	17.901	75.988 a	33.143
Total Manganese *	3866.667 a	1799.527	1279.778 b	515.208	853.445 b	220.892	1251.222 b	527.411	4202.222 a	1314.047
Total Zinc *	108.689 a	23.117	77.433 b	13.647	78.308 b	22.034	68.633 b	17.801	75.987 b	33.143
Nitrate **	13.187 a	15.761	8.201 a	11.419	18.846 a	32.899	7.8026 a	11.191	5.451 a	6.043
Ammonia *, **	9.799 a	4.712	26.608 bc	15.447	32.782 b	20.947	16.134ac	9.157	11.958 a	5.454
Loss Upon Ignition *, **	20.667 a	10.199	32.667b	14.422	36.667 b	11.180	33.778 b	13.617	14.622 a	6.503
Paste pH *	6.76 a	0.18	6.42 b	0.11	6.51 b	0.05	6.49 b	0.11	6.53 a	.16
Sulphate *, **	113.381 b	70.499	111.653 b	36.761	189.256 a	70.045	92.901 bc	32.585	49.182 c	18.325
WAD Cyanide	BDL		BDL		BDL		BDL		BDL	
Total Cyanide	BDL		BDL		BDL		BDL		BDL	
Thiocyanate	BDL		BDL		BDL		BDL		BDL	

6.3. Graphical Analysis of the Common Variable Data Set.

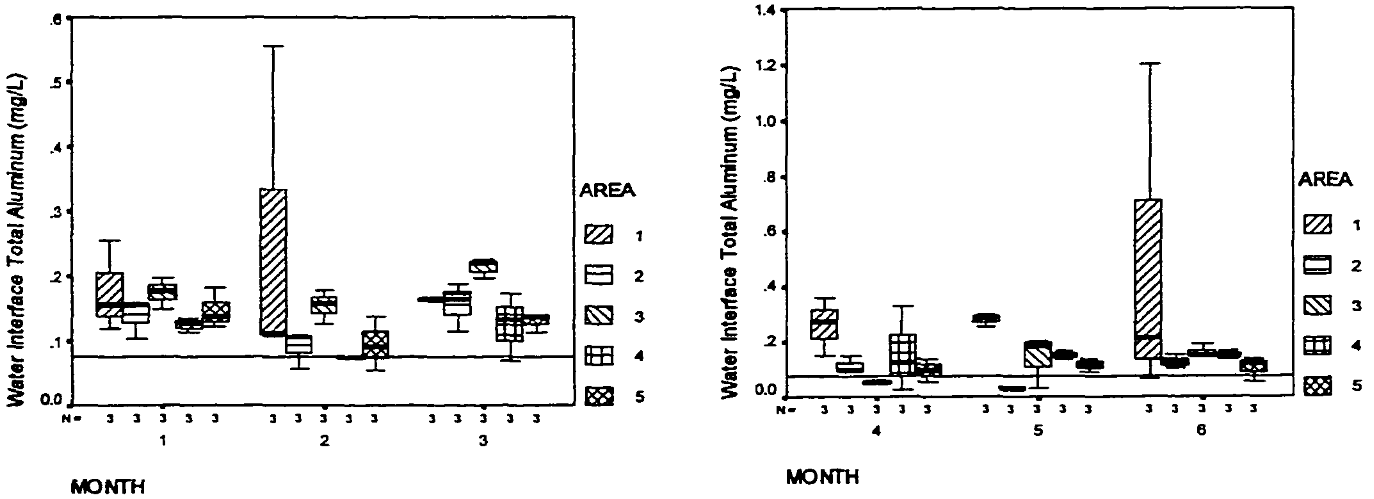
Boxplots of total aluminum, total copper, total iron, total manganese, total zinc, nitrate, pH, and sulphate concentrations (Figures 6.3.1. a- f. - Figures 6.3.8. a- f) were produced and analyzed for similarities in monthly mean concentration between Sample Area 1 and the Control Area in order to determine which of the common variable data set parameters would be further included in the discriminant function and fitting analyses to follow. Of the common variable data set parameters, total manganese consecutively showed high mean sediment concentrations at these areas over both years (Figures 6.3.4 e,f), and it was decided that this parameter would make a poor candidate for inclusion in the remaining analyses. Total manganese was therefore removed from the common variable data set at this step. Following this parameter's removal, the common variable data set became the reduced common variable data set.

Figures 6.3.1. a,b,c,d,e,f Boxplots of monthly mean Total Aluminum concentrations at Sampling Areas. 0=Discharge Channel, 5= Control. Solid reference line indicates an Ontario PWQO or PSQG.

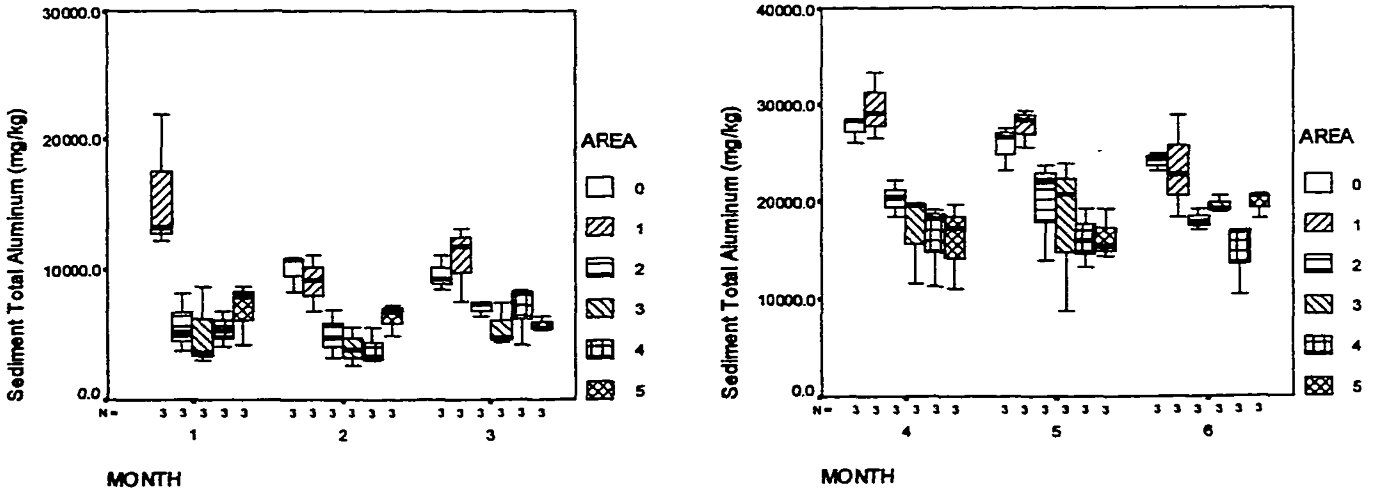
Figures 6.3.1.a,b; 1997 Surface Water (Right), 1998 Surface Water (Left).



Figures 6.3.1. c,d; 1997 Sed/ Water Interface (Right), 1998 Sed/Water Interface (Left).

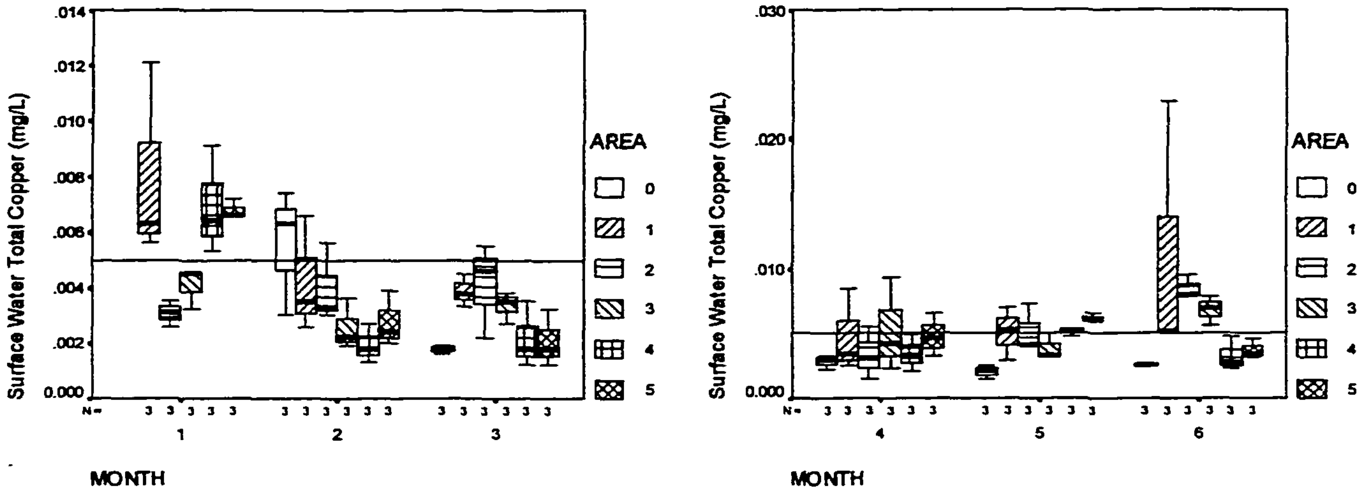


Figures 6.3.1 e,f; 1997 Lake Sediment (Right), 1998 Lake Sediment (Left).

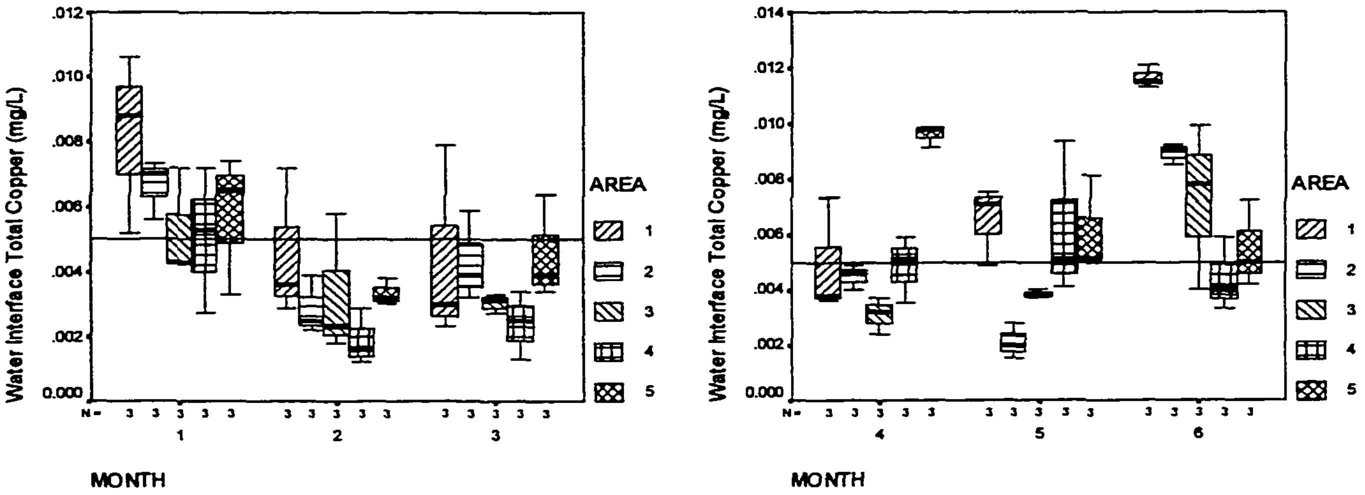


Figures 6.3.2. a,b,c,d,e,f Boxplots of monthly mean Total Copper concentrations at Sampling Areas. 0=Discharge Channel, 5= Control. Solid reference line indicates an Ontario PWQO or PSQG.

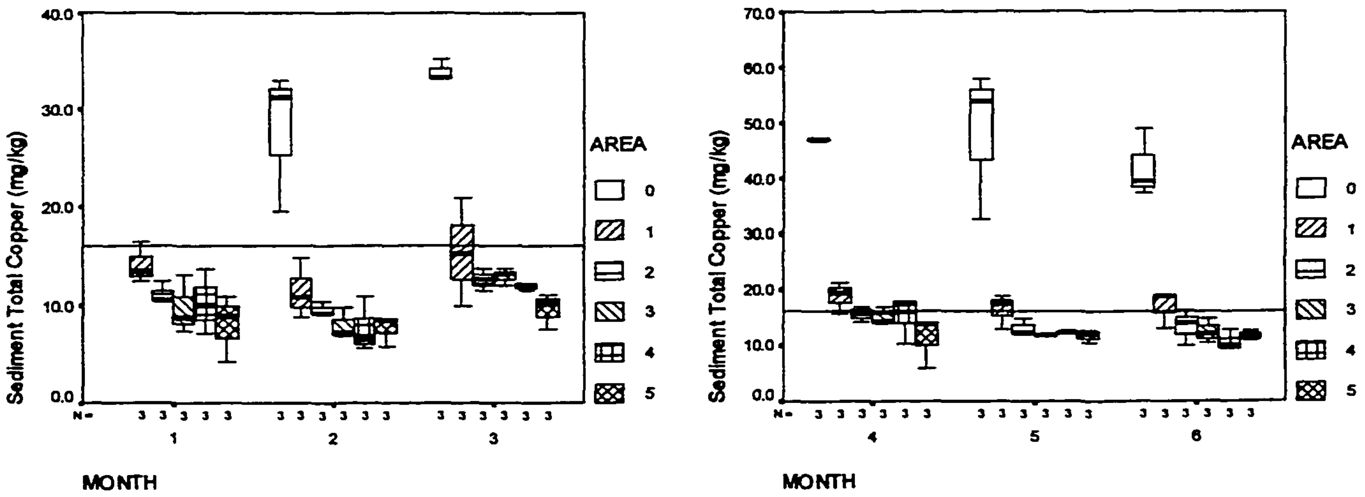
Figures 6.3.2.a,b; 1997 Surface Water (Right), 1998 Surface Water (Left).



Figures 6.3.2. c,d; 1997 Sed/ Water Interface (Right), 1998 Sed/Water Interface (Left).

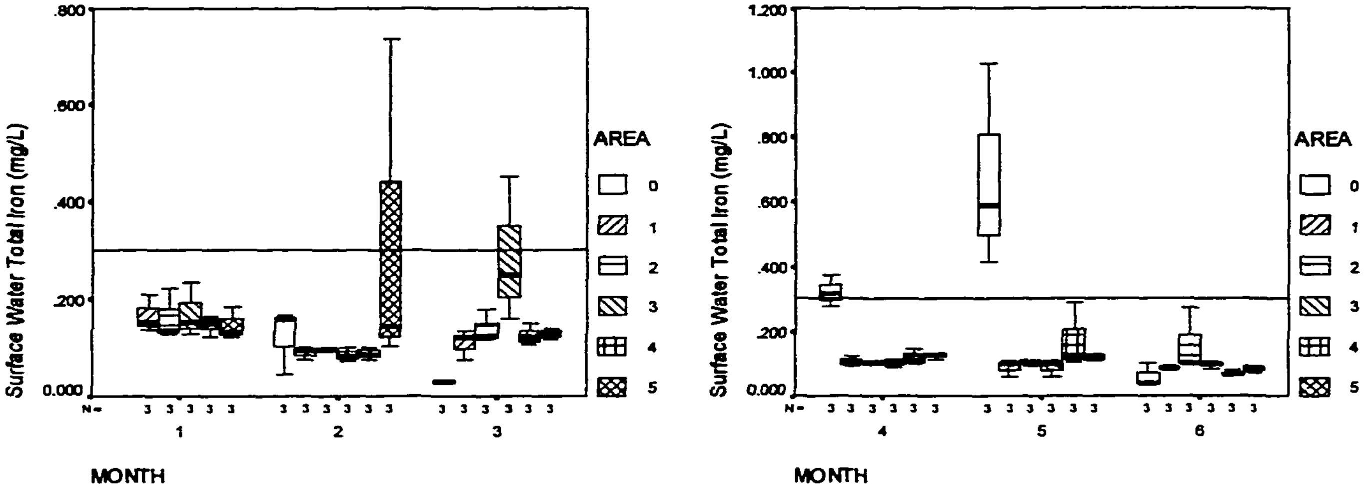


Figures 6.3.2 e,f; 1997 Lake Sediment (Right), 1998 Lake Sediment (Left).

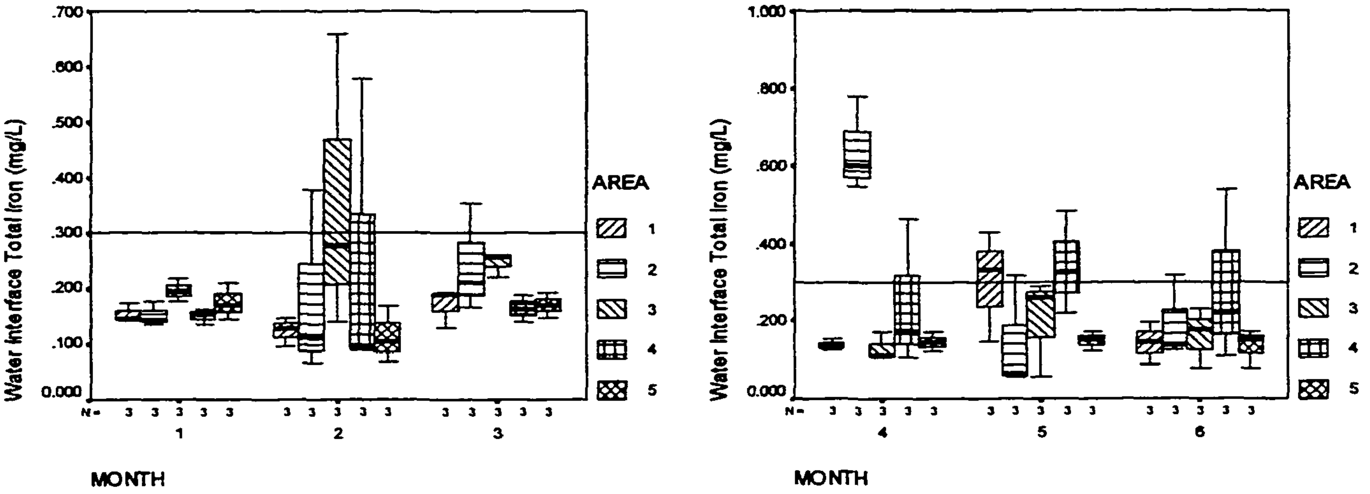


Figures 6.3.3. a,b,c,d,e,f Boxplots of monthly mean Total Iron concentrations at Sampling Areas. 0=Discharge Channel, 5= Control. Solid reference line indicates an Ontario PWQO or PSQG.

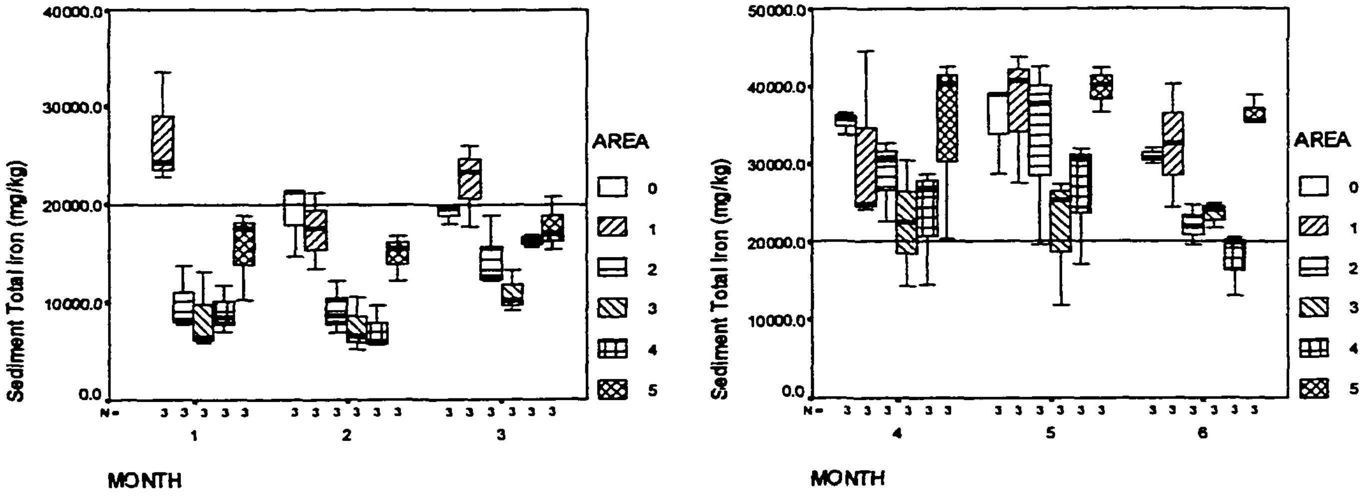
Figures 6.3.3.a,b; 1997 Surface Water (Right), 1998 Surface Water (Left).



Figures 6.3.3. c,d; 1997 Sed/ Water Interface (Right), 1998 Sed/Water Interface (Left).

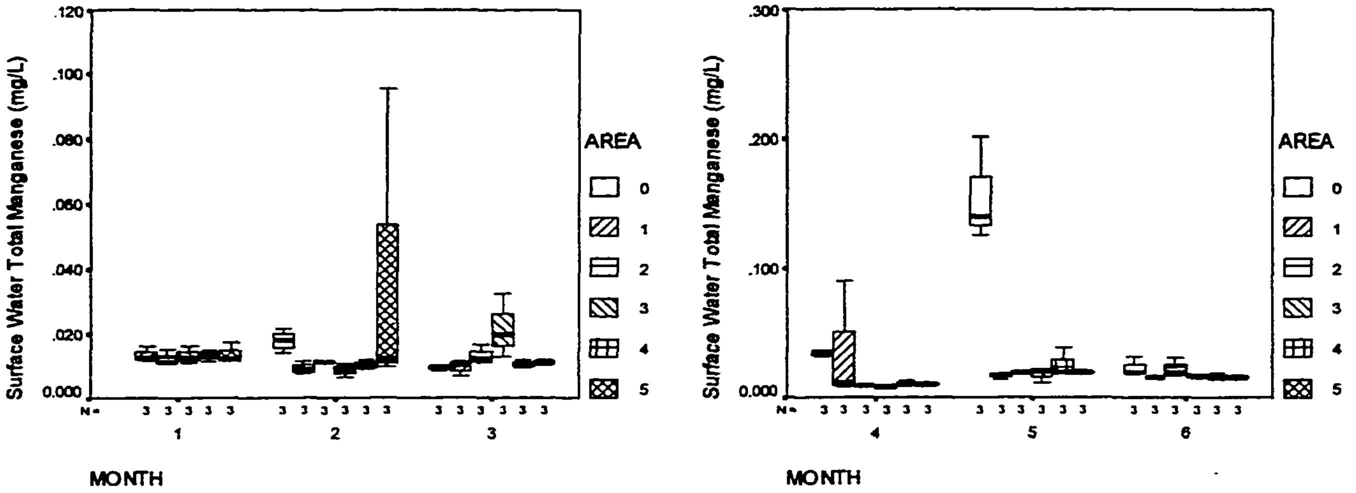


Figures 6.3.3 e,f; 1997 Lake Sediment (Right), 1998 Lake Sediment (Left).

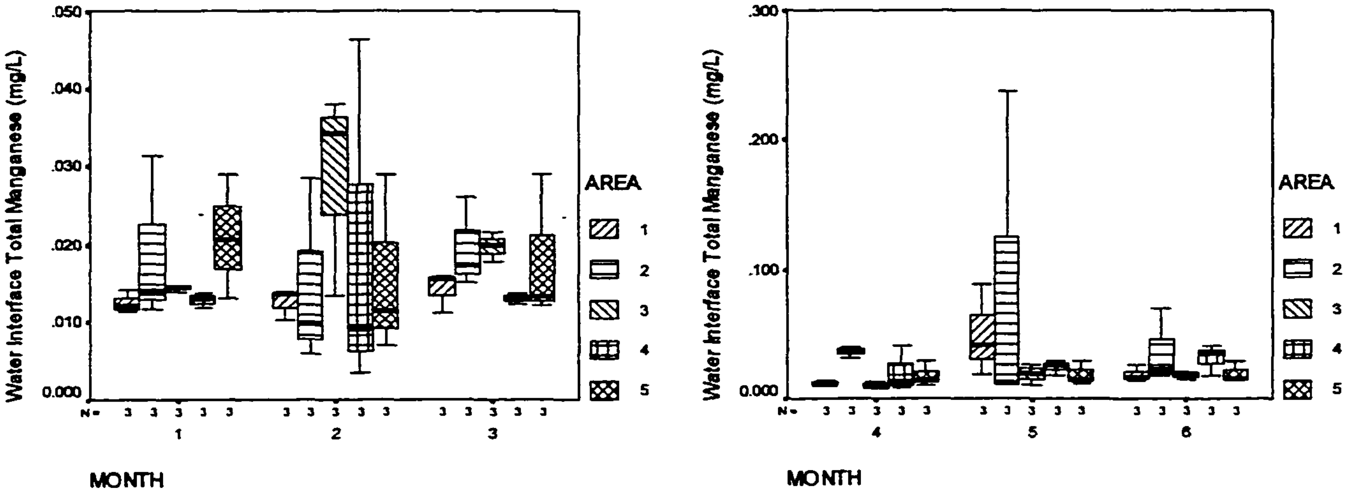


Figures 6.3.4. a,b,c,d,e,f Boxplots of monthly mean Total Manganese concentrations at Sampling Areas. 0=Discharge Channel, 5= Control. Solid or dashed reference line indicative of Ontario PWQO or PSQG.

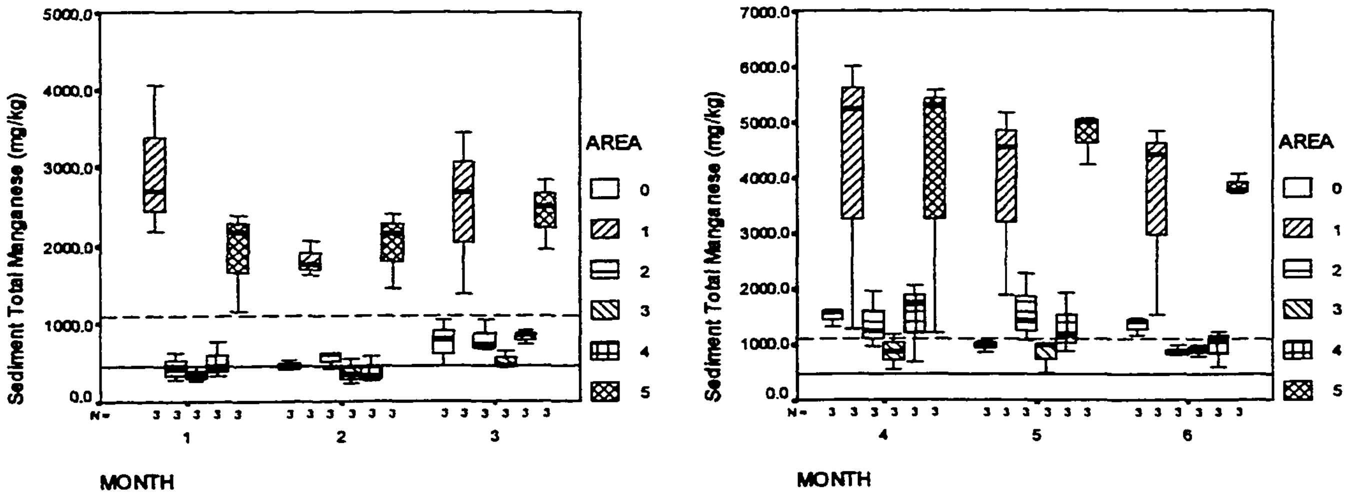
Figures 6.3.4. a,b; 1997 *Surface Water* (Right), 1998 *Surface Water* (Left).



Figures 6.3.4. c,d; 1997 Sed/ Water Interface (Right), 1998 Sed/Water Interface (Left).

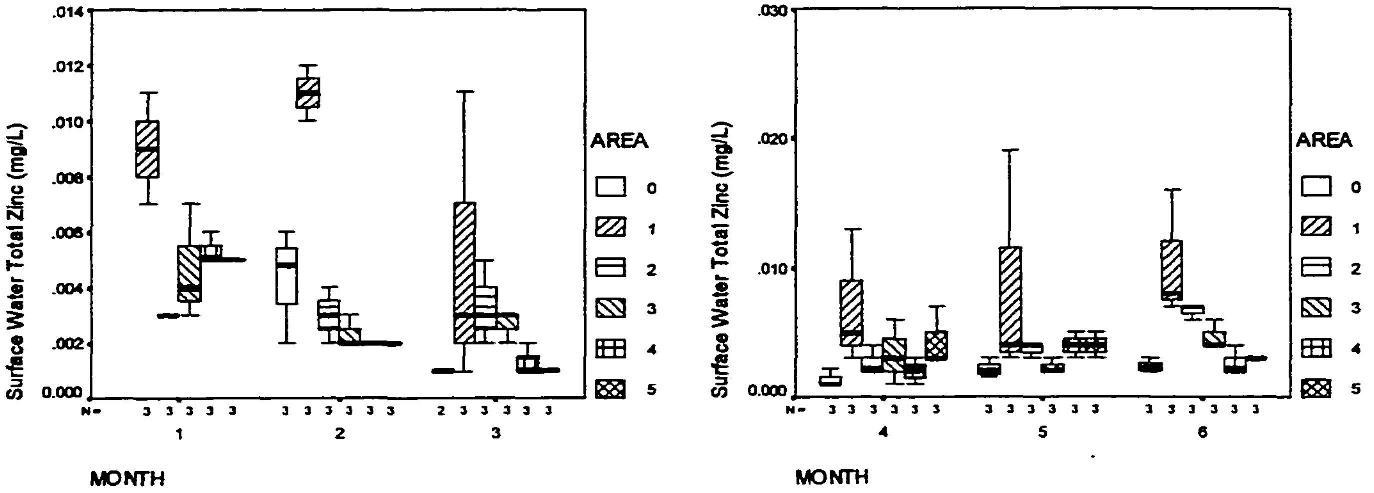


Figures 6.3.4 . e,f; 1997 Lake Sediment (Right), 1998 Lake Sediment (Left).

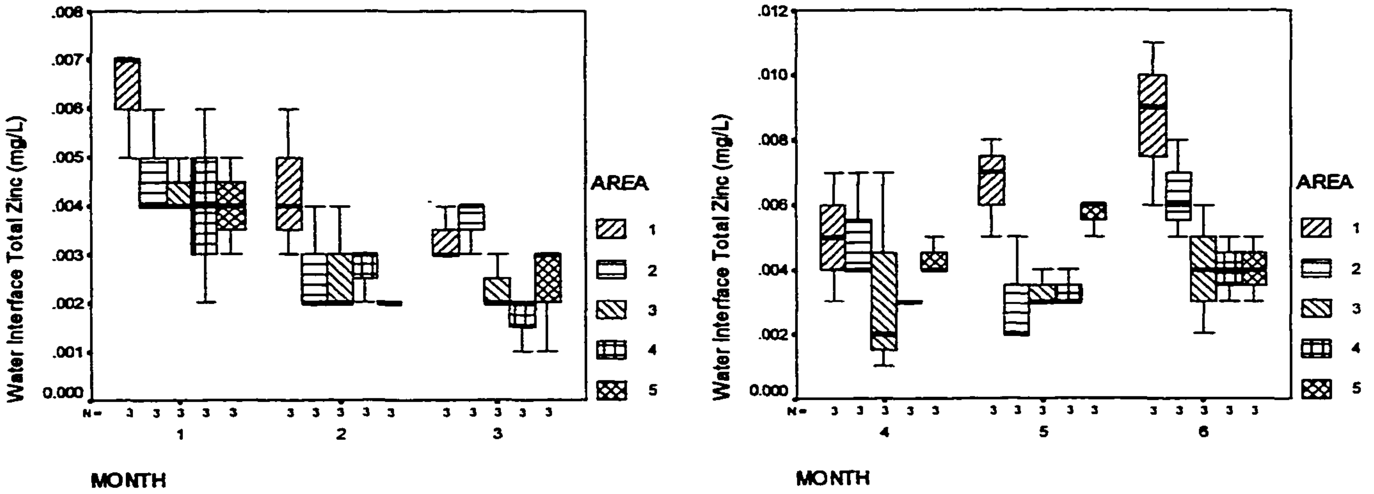


Figures 6.3.5. a,b,c,d,e,f Boxplots of monthly mean Total Zinc concentrations at Sampling Areas. 0=Discharge Channel, 5= Control. Solid reference line indicates an Ontario PWQO or PSQG.

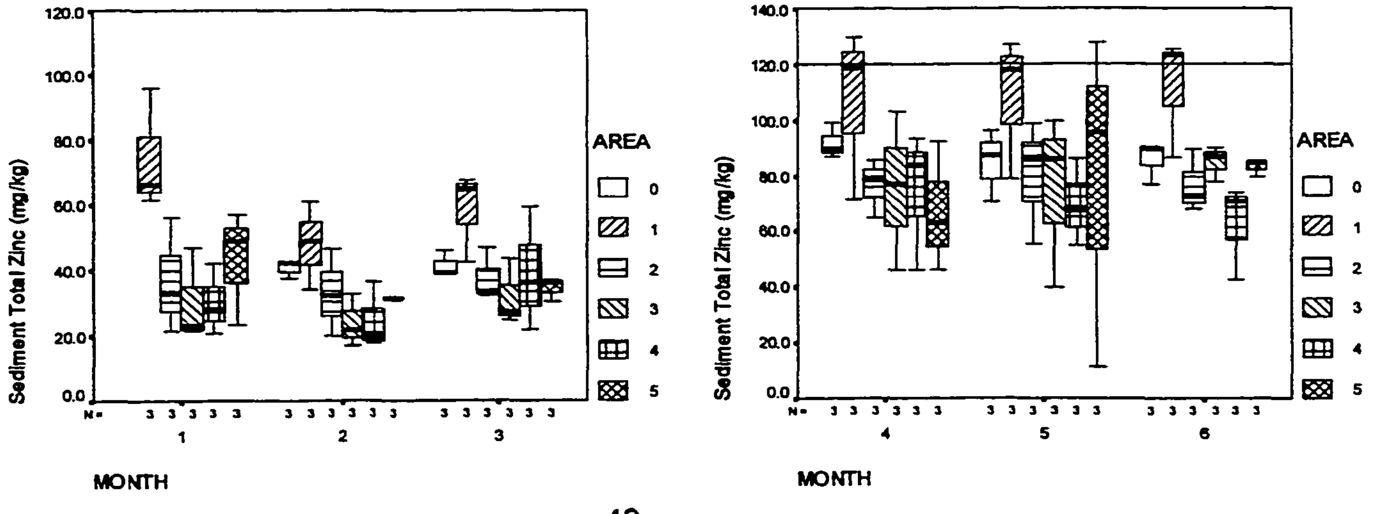
Figures 6.3.5. a,b; 1997 Surface Water (Right), 1998 Surface Water (Left).



Figures 6.3.5. c,d; 1997 Sed/ Water Interface (Right), 1998 Sed/Water Interface (Left).

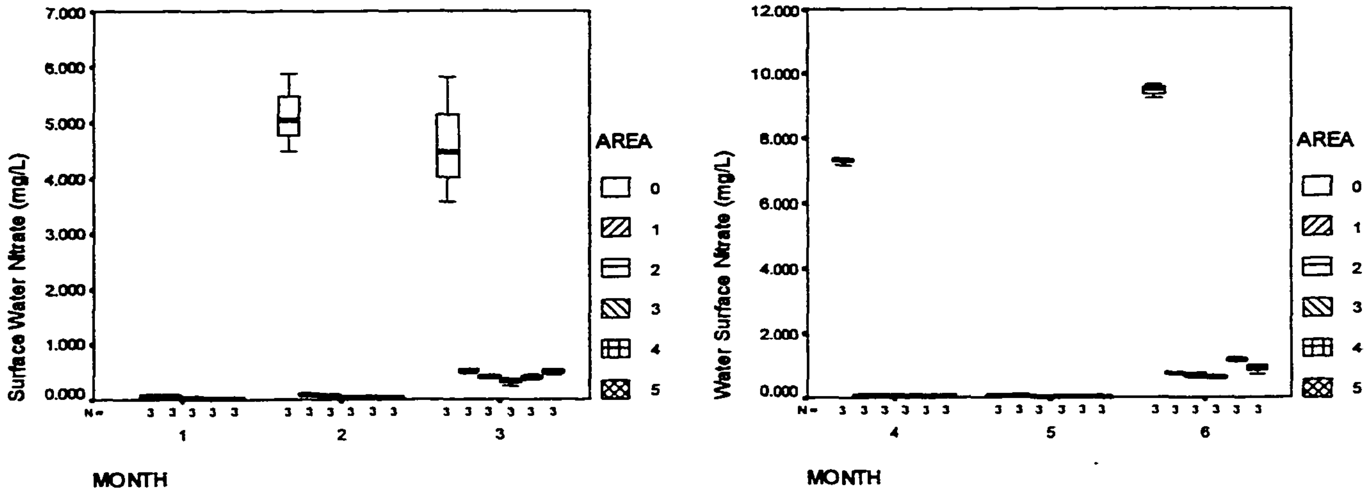


Figures 6.3.5. e,f; 1997 Lake Sediment (Right), 1998 Lake Sediment (Left).

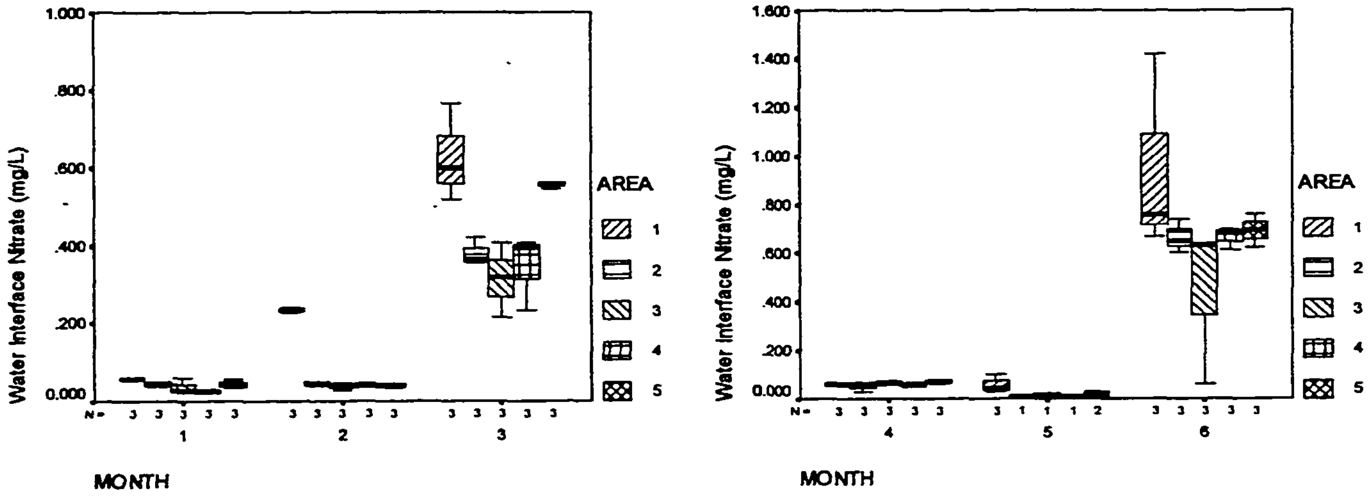


Figures 6.3.6. a,b,c,d,e,f Boxplots of monthly mean Nitrate concentrations at Sampling Areas. 0=Discharge Channel, 5= Control.

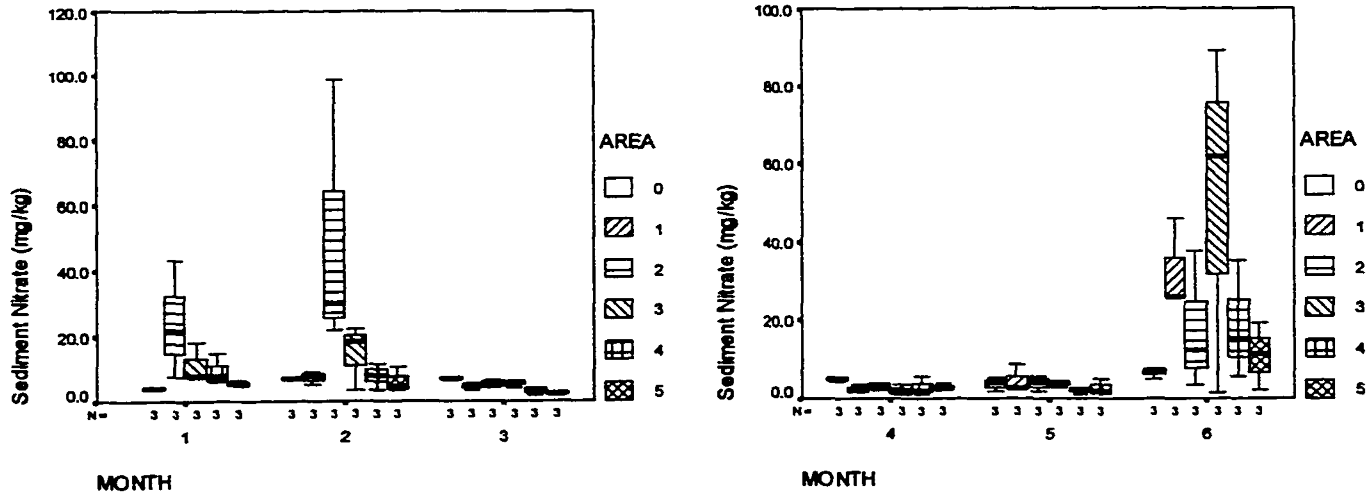
Figures 6.3.6. a,b; 1997 Surface Water (Right), 1998 Surface Water (Left).



Figures 6.3.6. c,d; 1997 Sed/ Water Interface (Right), 1998 Sed/Water Interface (Left).

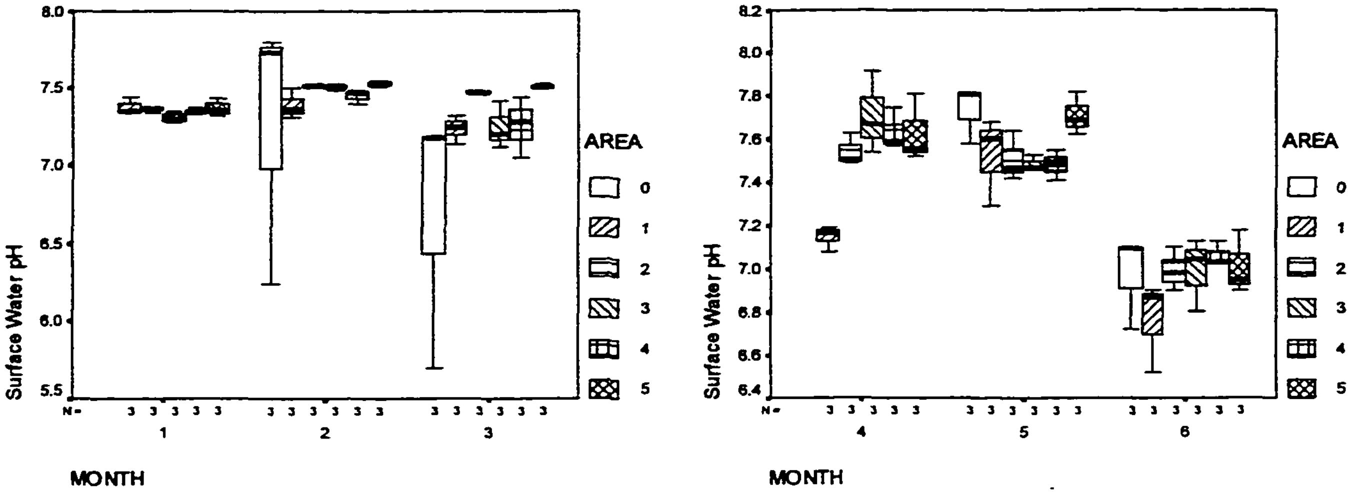


Figures 6.3.6. e,f; 1997 Lake Sediment (Right), 1998 Lake Sediment (Left).

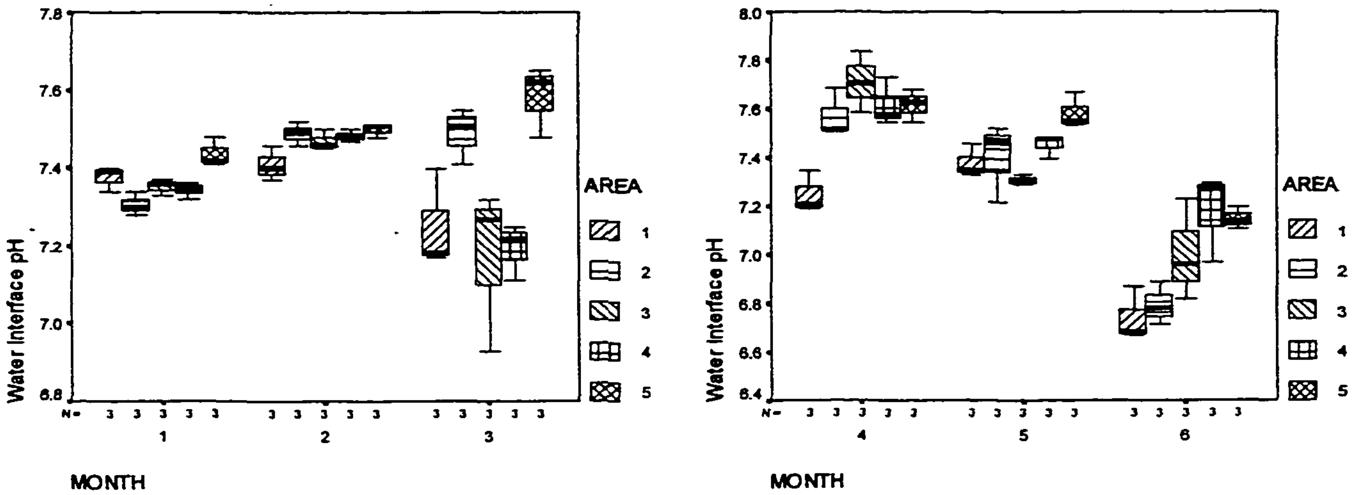


Figures 6.3.7. a,b,c,d,e,f Boxplots of monthly mean pH levels at Sampling Areas. 0=Discharge, 5=Control.

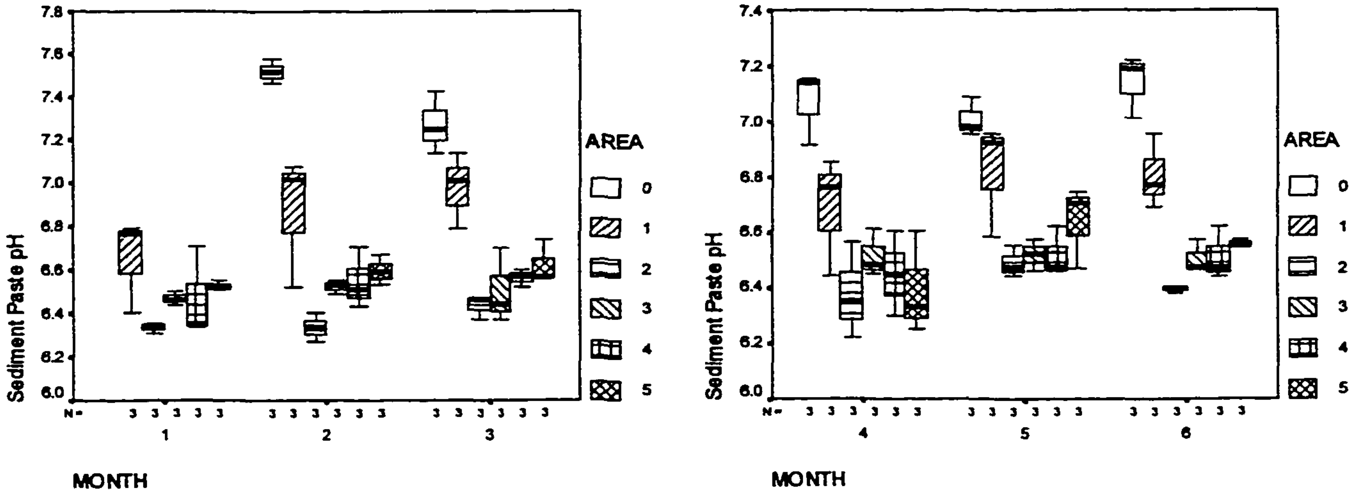
Figures 6.3.7. a,b; 1997 Surface Water (Right), 1998 Surface Water (Left).



Figures 6.3.7. c,d; 1997 Sed/ Water Interface (Right), 1998 Sed/Water Interface (Left).

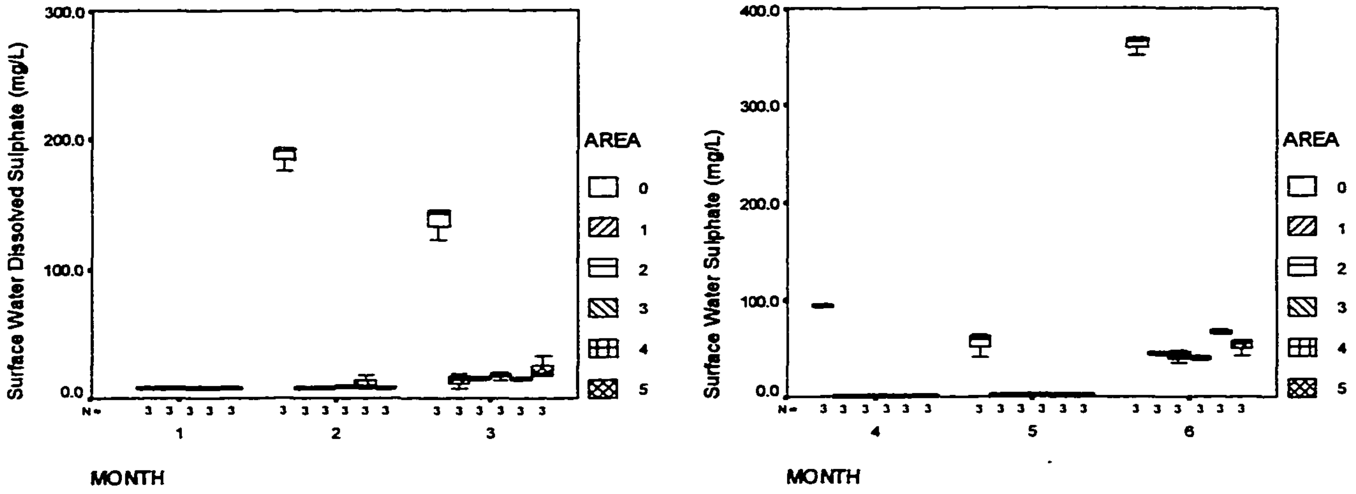


Figures 6.3.7. e,f; 1997 Lake Sediment (Right), 1998 Lake Sediment (Left).

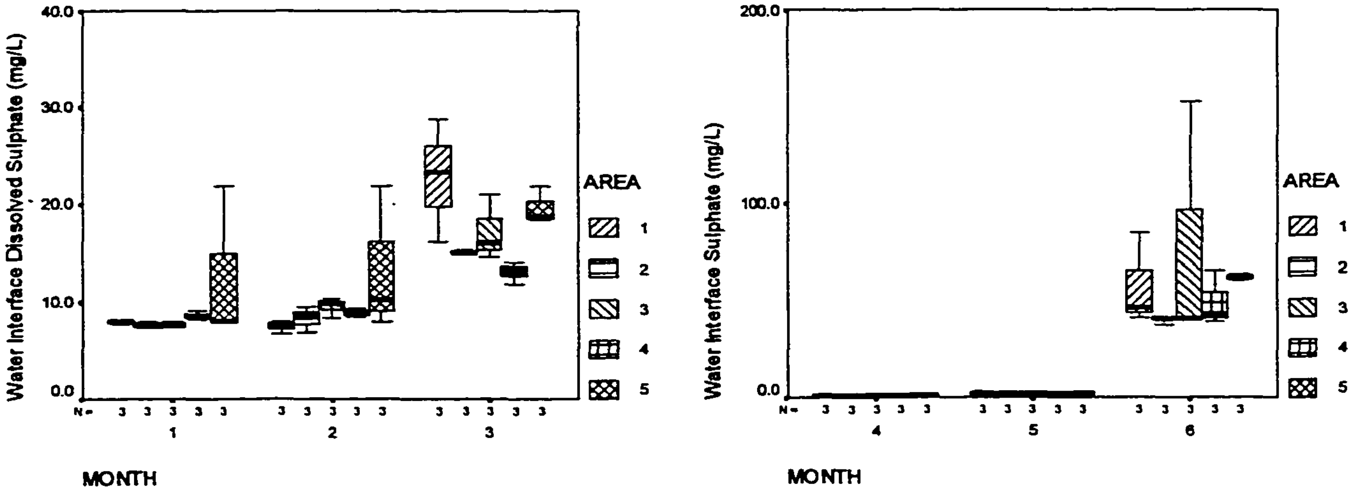


Figures 6.3.8. a,b,c,d,e,f Boxplots of monthly mean Sulphate concentrations at Sampling Areas. 0=Discharge Channel, 5= Control.

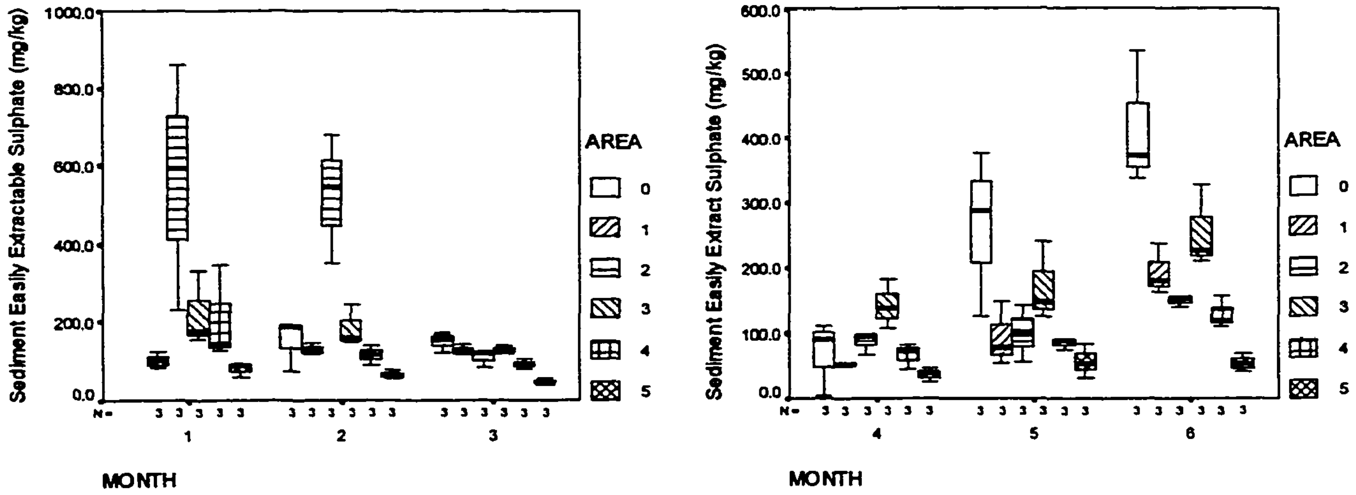
Figures 6.3.8. a,b; 1997 Surface Water (Right), 1998 Surface Water (Left).



Figures 6.3.8. c,d; 1997 Sed/ Water Interface (Right), 1998 Sed/Water Interface (Left).



Figures 6.3.8. e,f; 1997 Lake Sediment (Right), 1998 Lake Sediment (Left).



6.4 Parameters of Importance

Parameters of importance in this study were identified by meeting each of the following three criteria; 1) being significantly higher in concentration at Sample Area 1 than at other lake areas as determined through ANOVA and LSD testing, 2) showing concentrations either within lake water or lake sediment that were above published provincial guidelines (MOEE, 1994), and 3) that demonstrated concentrations within the mines's effluent approximately equal to or higher than those observed in the receiving lake.

6.5 Discriminant Analysis;

Discriminant analysis using the reduced common variable data set, successfully classified those sampling sites belonging to Sample Area 1 and the Control Area with the greatest accuracy at every lake level over both years (Table 6.5.1). Of the discriminant functions generated, the first function consistently accounted for a significant proportion of the "among group" variation (Tables 6.5.2., 6.5.3., & 6.5.4.). Distances "between group" centroids, illustrated graphically in Figures 6.5.1-3, reveal that three major groups of sampling sites existed; those belonging to Sample Area 1, the Control Area, and the remaining lake areas.

Sampling sites belonging to area 1 typically separated from other lake sites at the positive end of the first discriminant function axis on those functions used to describe lake sediment (Figures 6.5.3. a,b) and the 1997 lake surface water (Figure 6.5.1. a). Concentrations of total aluminum, total copper, total iron, sulphate and pH levels were shown to be the most discriminating parameters at the positive ends of these functions for the 1997 and 1998 lake sediments (Tables 6.5.4. a,b). Total zinc concentration was found to be the most discriminating parameter at the positive end of the function used to describe 1997 surface water (Table 6.5.2 a).

The pattern of group separation and discriminating variable contribution was reversed for the sediment-water interface (Figures 6.5.2. a), and 1998 surface water (Figure 6.5.1. b) reduced common variable data set parameters. In these analyses, sites belonging to sample area 1 ranked in at the negative end of the first discriminant function axis while other lake sites scored relatively positively. Concentrations of the total aluminum and total zinc were found to be the most discriminating parameters at the negative end of these functions (Tables 6.5.2.b, 6.5.3.a, & 6.5.3.b).

Table 6.5.1 Summary of sites correctly grouped into respective sampling areas through discriminant function analysis of 1997 and 1998 reduced common variable data sets.

Year	Data Set	Percent of Area 1 sites correctly grouped	Percent of Area 2 sites correctly grouped	Percent of Area 3 sites correctly grouped	Percent of Area 4 sites correctly grouped	Percent of Control sites correctly grouped	Percent of all sites correctly grouped
1997	Surface Water	66.7	55.6	33.3	66.7	33.3	51.1
1997	Sed / Water Interface	77.8	55.6	66.7	66.7	77.8	68.9
1997	Sediment	77.8	66.7	66.7	55.6	100.0	73.3
1998	Surface Water	77.8	33.3	77.8	66.7	33.3	57.8
1998	Sed / Water Interface	77.8	71.4	42.9	57.1	62.5	63.2
1998	Sediment	88.9	55.6	77.8	66.7	100.0	77.8

Tables 6.5.2.a, b. Discriminant function data for 1997 and 1998 surface water reduced common variable data sets that includes the amount of variation explained by each function, the standardized canonical coefficient values assigned to each variable, and the pooled within-groups correlations between discriminating variables and standardized canonical discriminant functions.

6.5.2. a (1997 surface water)

	Function 1	Function 2		After Function 1	After Function 2
Eigen Value	1.920	0.368	Chi-square	62.234	21.516
% Var. Explained	75.0	14.4	df	28	18
Canonical Corr.	0.811	0.518	Wilks Lambda	0.194	0.568
Significance	0.000	0.254			
	Function 1	Function 2		Function 1	Function 2
Parameter	Standardized Coefficients			Pooled within-groups Correlations	
Total Aluminum	0.050	-0.325		0.042	-0.354
Total Copper	-0.267	0.578		0.236	0.145
Total Iron	-0.087	0.254		-0.120	0.149
Total Zinc	1.182	-0.018		0.727	-0.032
Nitrate	1.102	0.767		0.101	0.157
pH	-0.009	0.905		-0.200	0.731
Sulphate	-0.701	-0.223		-0.111	0.084

6.5.2. b (1998 surface water)

	Function 1	Function 2		After Function 1	After Function 2
Eigen Value	1.759	0.800	Chi-square	64.957	26.386
% Var. Explained	65.9	30.0	df	28	18
Canonical Corr.	0.799	0.667	Wilks Lambda	0.181	0.499
Significance	0.000	0.091			
	Function 1	Function 2		Function 1	Function 2
Parameter	Standardized Coefficients			Pooled within-groups Correlations	
Total Aluminum	-0.315	1.267		-0.214	0.362
Total Copper	0.495	0.167		-0.218	-0.089
Total Iron	0.596	0.284		0.132	0.194
Total Zinc	-0.992	-0.171		-0.548	0.052
Nitrate	-2.345	17.652		0.048	0.144
pH	1.453	0.296		0.251	-0.013
Sulphate	3.833	-17.205		0.043	0.117

Tables 6.5.4.a, b. Discriminant function data for 1997 and 1998 sediment reduced common variable data sets that includes the amount of variation explained by each function, the standardized canonical coefficient values assigned to each variable, and the pooled within-groups correlations between discriminating variables and standardized canonical discriminant functions.

6.5.4. a (1997 sediment)

	Function 1	Function 2		After Function 1	After Function 2
Eigen Value	5.869	1.172	Chi-square	126.808	53.581
% Var. Explained	74.1	14.8	df	28	18
Canonical Corr.	0.924	0.735	Wilks Lambda	0.036	0.244
Significance	0.000	0.000			
	Function 1	Function 2		Function 1	Function 2
Parameter	Standardized Coefficients			Pooled within-groups Correlations	
Total Aluminum	0.425	1.710		0.351	0.517
Total Copper	-1.372	0.768		0.088	0.567
Total Iron	1.108	-1.497		0.467	0.238
Total Zinc	0.047	-0.359		0.297	0.437
Nitrate	0.022	0.160		-0.181	0.097
pH	0.895	0.437		0.437	0.359
Sulphate	-0.199	0.084		-0.269	0.252

6.5.4. b (1998 sediment)

	Function 1	Function 2		After Function 1	After Function 2
Eigen Value	4.300	2.311	Chi-square	121.692	58.320
% Var. Explained	61.6	33.1	df	28	18
Canonical Corr.	0.901	0.835	Wilks Lambda	0.041	0.216
Significance	0.000	0.000			
	Function 1	Function 2		Function 1	Function 2
Parameter	Standardized Coefficients			Pooled within-groups Correlations	
Total Aluminum	0.923	0.384		0.371	0.466
Total Copper	0.562	0.054		0.333	0.250
Total Iron	-1.317	0.503		-0.120	0.480
Total Zinc	0.251	-0.234		0.215	0.290
Nitrate	-0.233	0.680		0.103	-0.066
pH	0.388	0.652		0.272	0.488
Sulphate	0.552	-0.925		0.321	-0.390

Figures 6.5.1 a,b Discriminant funciton plots for 1997 and 1998 Surface Water, Reduced Common Variable Data Set Parameters. Groups 1 to 4 represent sampling sites identified as belonging to Sampling Areas 1 to 4 respectively. Group 5 is representative of sites belonging to Control Area.

Figure 6.5.1 a

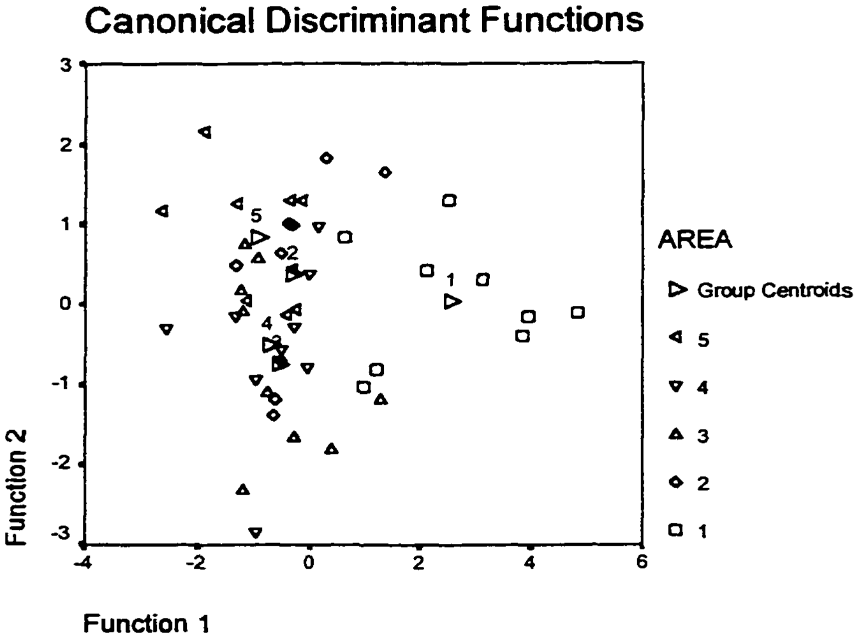
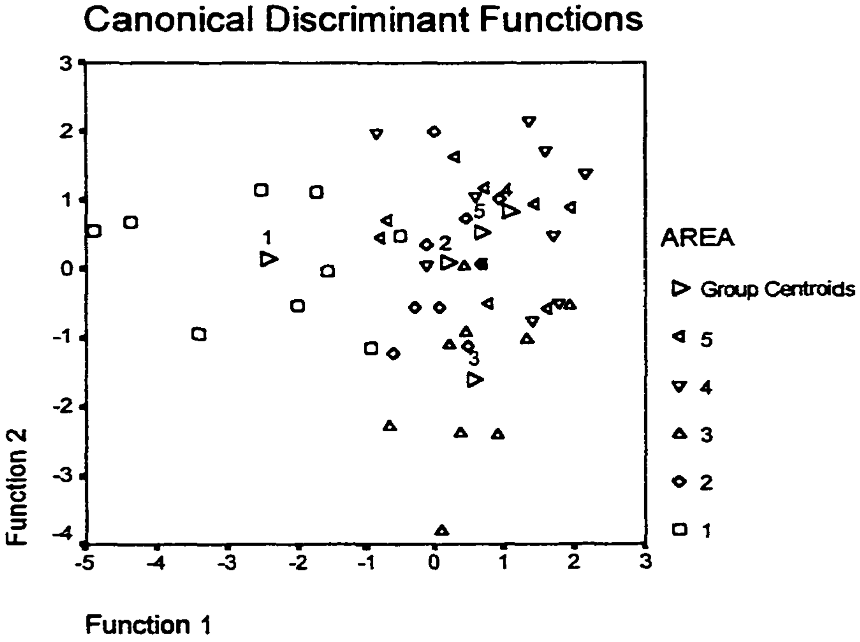


Figure 6.5.1. b



Figures 6.5.2 a,b Discriminant funciton plots for 1997 and 1998 Water / Sediment Interface, Reduced Common Variable Data Set Parameters. Groups 1 to 4 represent sampling sites identified as belonging to Sampling Areas 1 to 4 respectively. Group 5 is representative of sites belonging to Control Area.

Figure 6.5.2 a

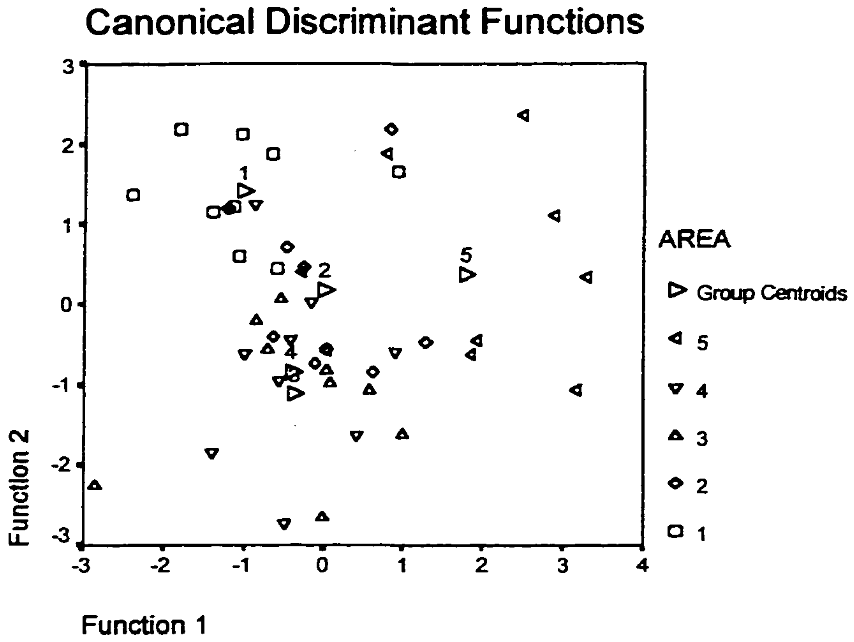
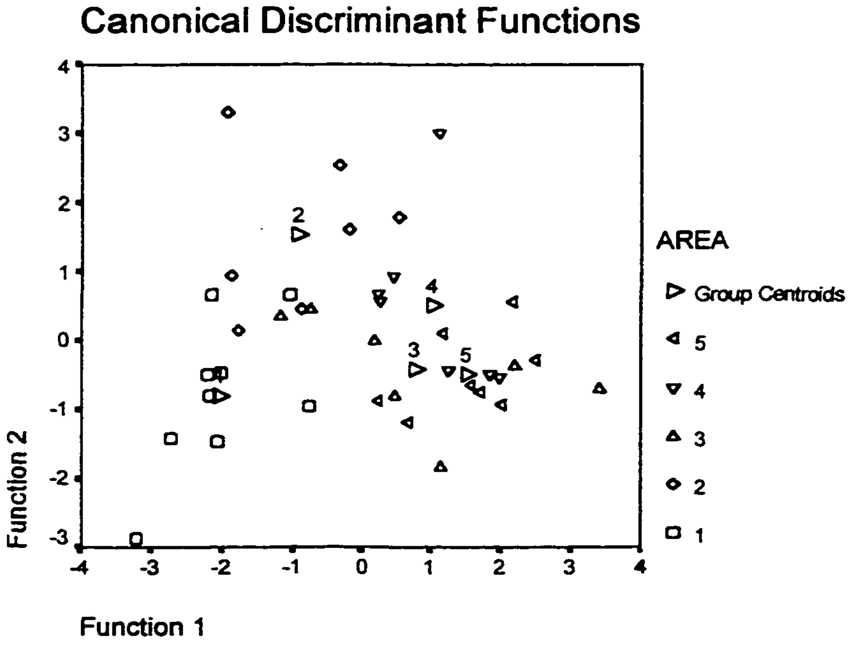


Figure 6.5.2. b



Figures 6.5.3 a,b Discriminant funciton plots for 1997 and 1998 Sediment, Reduced Common Variable Data Set Parameters. Groups 1 to 4 represent sampling sites identified as belonging to Sampling Areas 1 to 4 respectively. Group 5 is representative of sites belonging to Control Area.

Figure 6.5.3 a

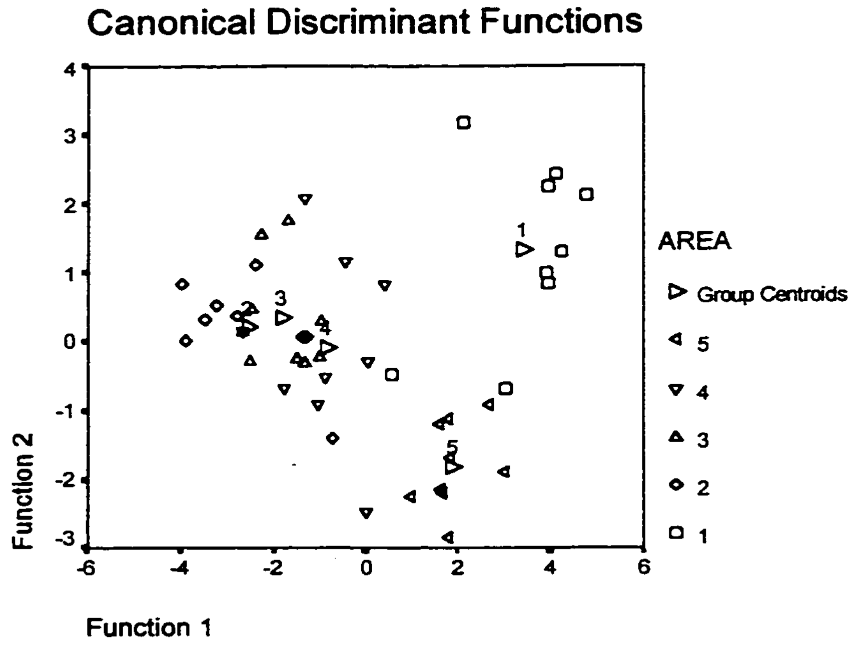
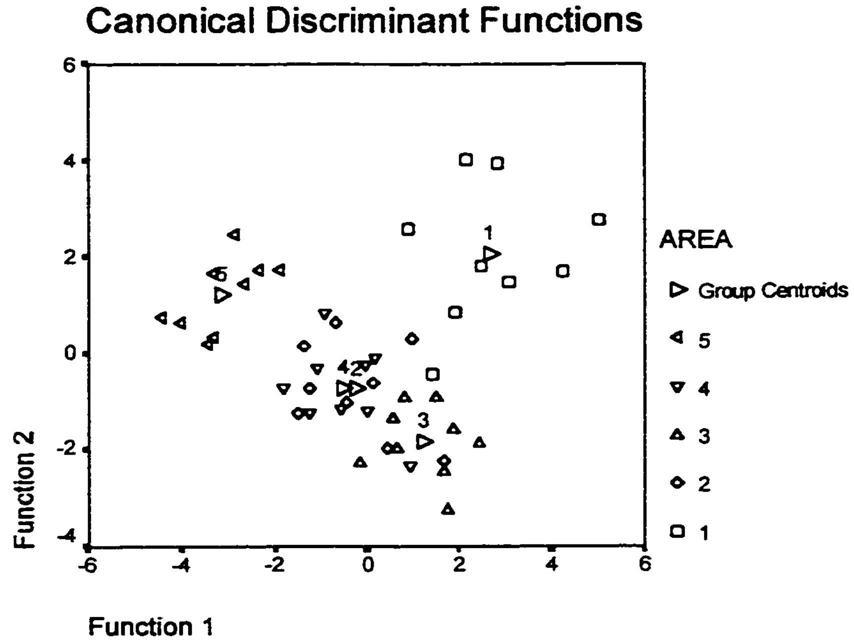


Figure 6.5.3 b



6.6 Regression Results

Regression analysis was used in this study to relate predicted effluent concentration to yearly mean, sign transformed, discriminant function score at sample sites on Lake 282. Linear regressions were performed for each common variable data set. Of the regression equations generated, the highest correlations with predicted lake effluent concentrations (Table 6.6.1.) were achieved after taking the natural log of the mean percent effluent data. The fitting functions generated in this analysis, illustrated graphically in Figures 6.6.1-3, were significant for all data sets but the one used to describe 1997 lake sediment (Figure 6.6.3.a).

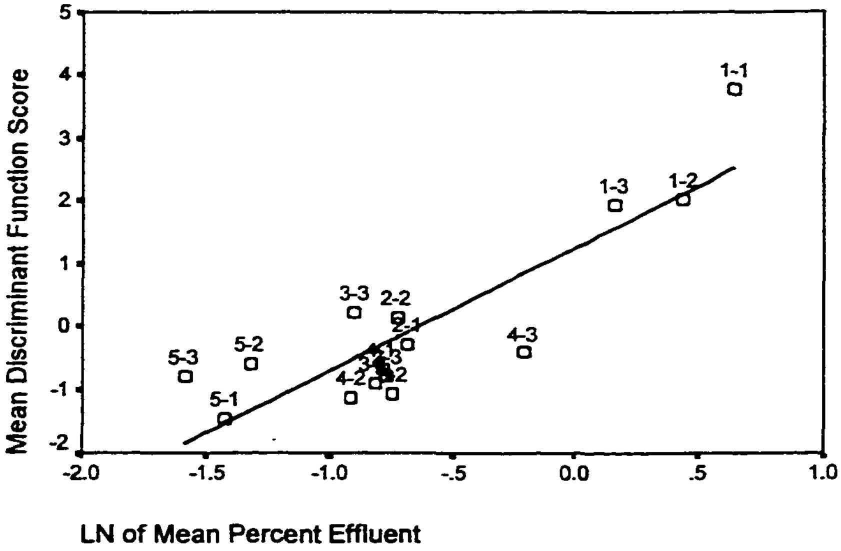
Table 6.6.1. Linear fitting functions used to predict discriminant function score from mean percent effluent in 1997 and 1998.

					Constants .	
Reduced Common Variable Data Set	R sq	df	F	sig F	B0	B1
Surface Water 1997	0.748	13	38.64	0.000	1.2570	1.9529
Surface Water 1998*	0.611	13	20.39	0.001	1.1451	1.7790
Sediment / Water Interface 1997*	0.521	13	14.15	0.002	0.7617	1.1834
Sediment / Water Interface 1998*	0.650	13	24.10	0.000	1.0966	1.7036
Sediment 1997	0.125	13	1.87	0.195	0.8527	1.3248
Sediment 1998	0.694	13	29.52	0.000	1.7210	2.6737

*sign transformed functions

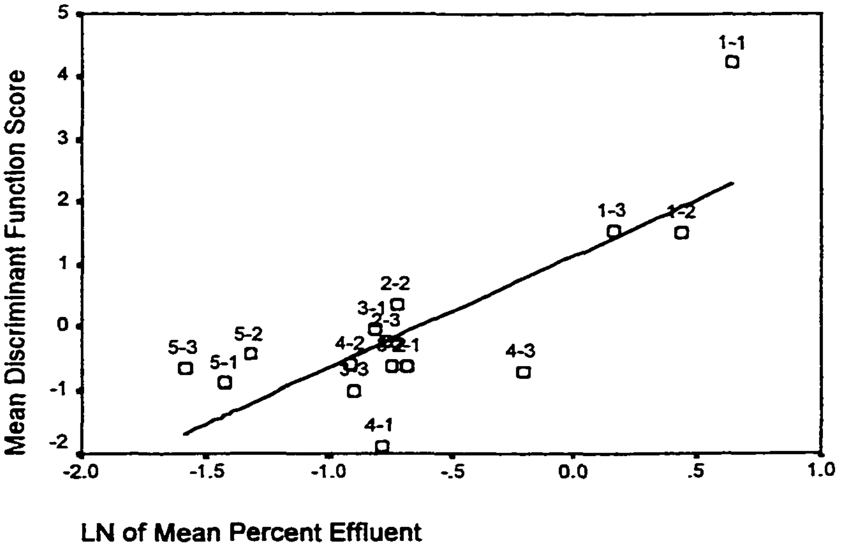
Figures 6.6.1.a,b. Plots of Mean Discriminant Function Score vs the Natural Log of Mean Percent Effluent for 1997 and 1998 Surface Water at Lake 282 Sampling Sites.

Figure 6.6.1. a



Mean Discriminant Function Score vs Natural Log of Mean Percent Effluent for 1997 Surface Water Data

Figure 6.6.1. b



Mean Discriminant Function Score vs Natural Log of Mean Percent Effluent for 1998 Surface Water Data

Figures 6.6.2.a,b.. Plots of Mean Discriminant Function Score vs the Natural Log of Mean Percent Effluent for 1997 and 1998 Sediment / Water Interface at Lake 282 Sampling Sites.

Figure 6.6.2. a

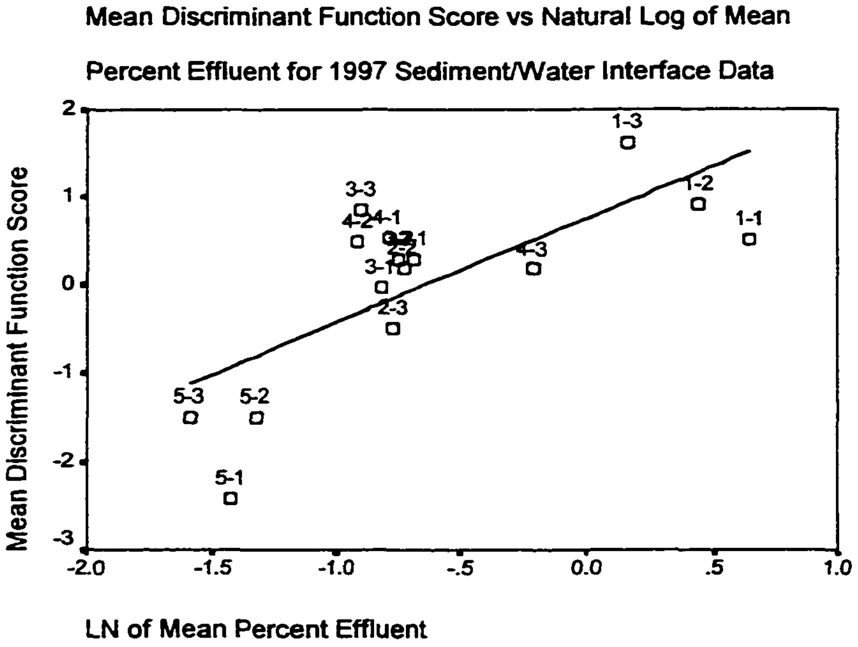
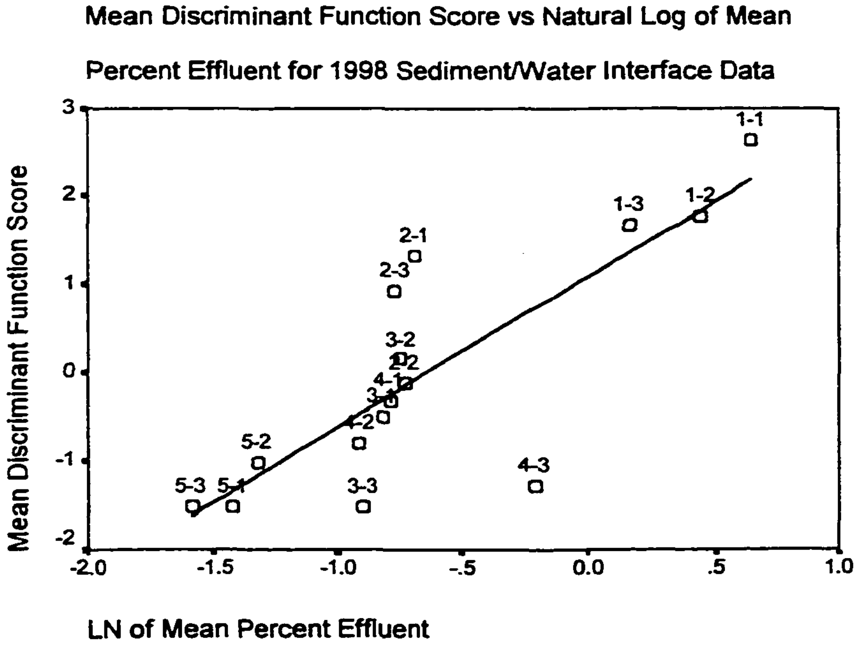


Figure 6.6.2. b



Figures 6.6.3.a,b. Plots of Mean Discriminant Function Score vs the Natural Log of Mean Percent Effluent for 1997 and 1998 Sediment at Lake 282 Sampling Sites.

Figure 6.6.3. a

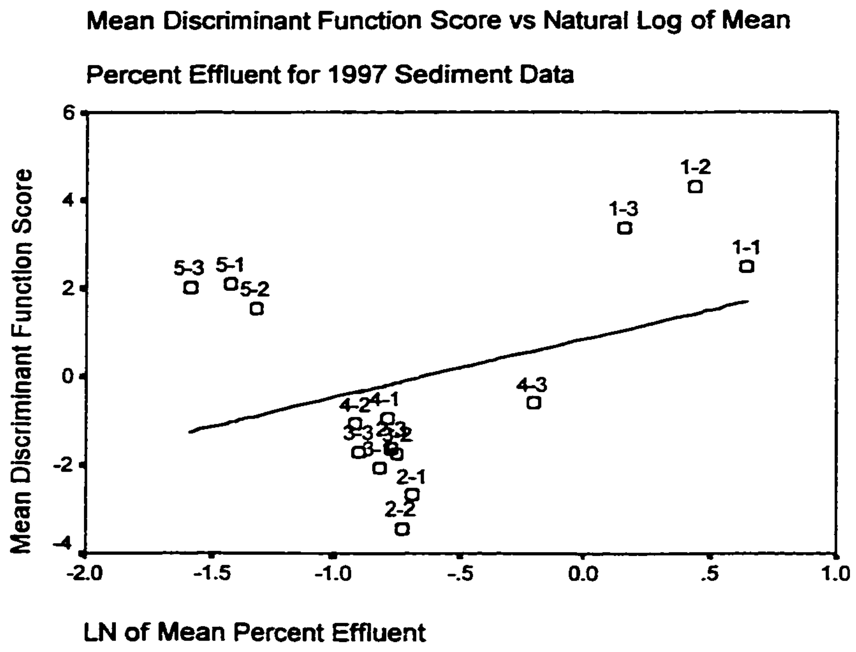
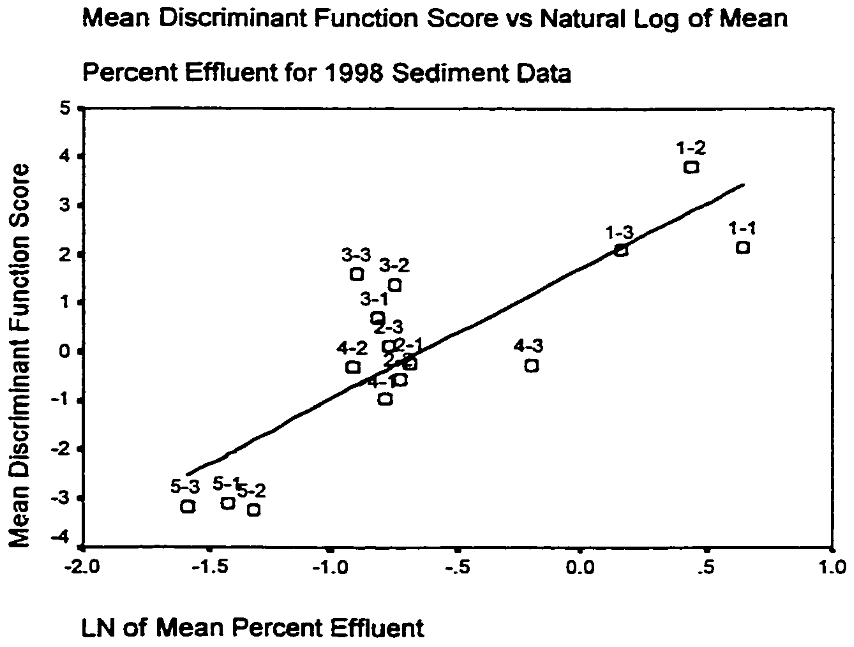


Figure 6.6.3. b



6.7 Plume transformations and predicted discriminant function plots.

Statistically significant fitting equations obtained from the regression analyses were used to transform the 1998 mean percent effluent plot in order to generate a predictive discriminant function plot for each of the surface (Figure 6.7.1), sediment/water interface (Figure 6.7.2), and sediment (Figure 6.7.3) lake levels for each year of study. These discriminant function plots consecutively showed in each data set, a lake region encompassing Sample Area 1 in which the discriminant function score was predicted to be from between 2 to 4.5 times greater than any other sample area.

Figure 6.7.1. Plots of Predicted Discriminant Function Scores for 1997 and 1998 Lake Surface Water.

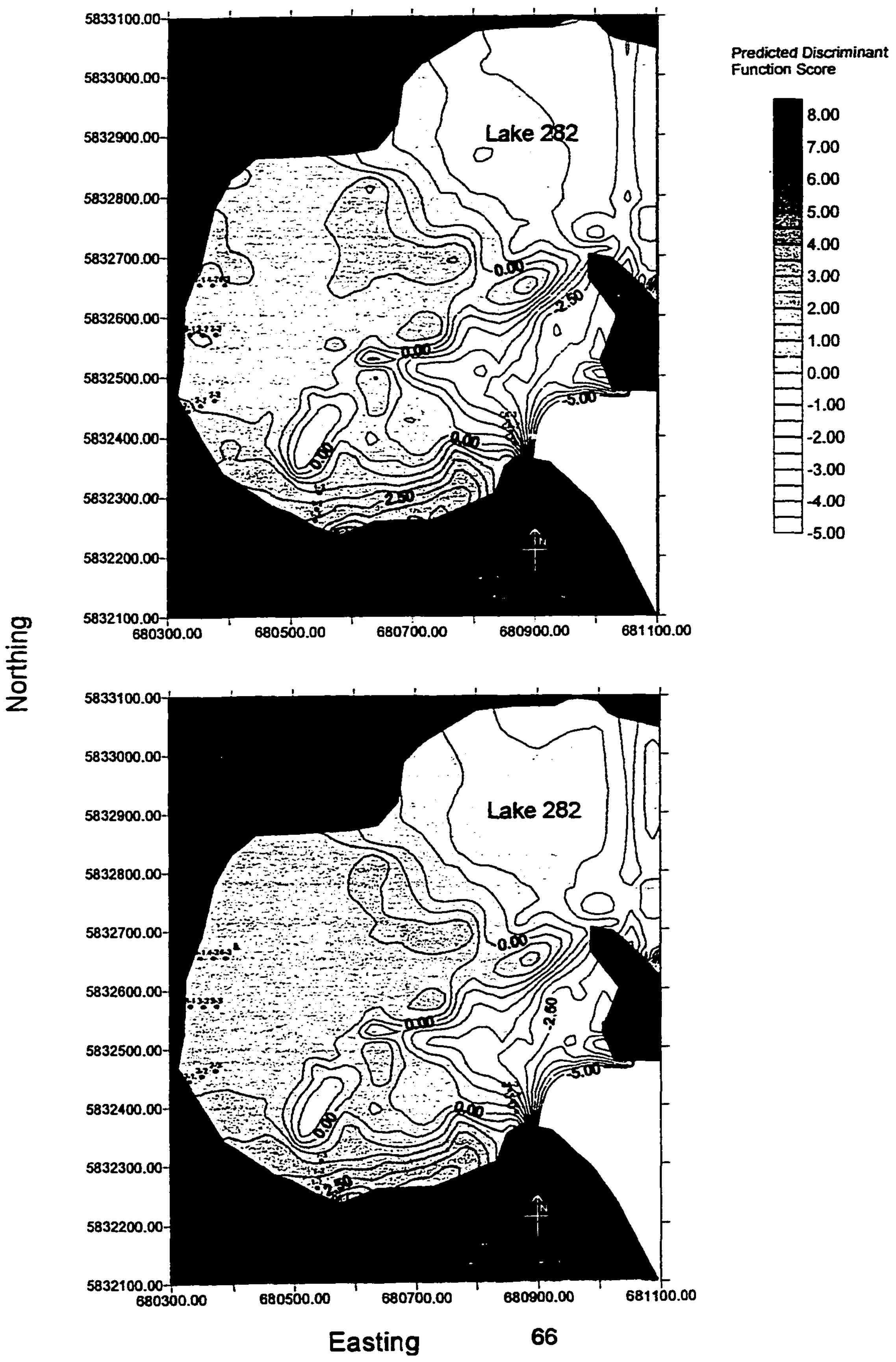


Figure 6.7.2. Plots of Predicted Discriminant Function Scores for 1997 and 1998 Lake Sediment / Water Interface.

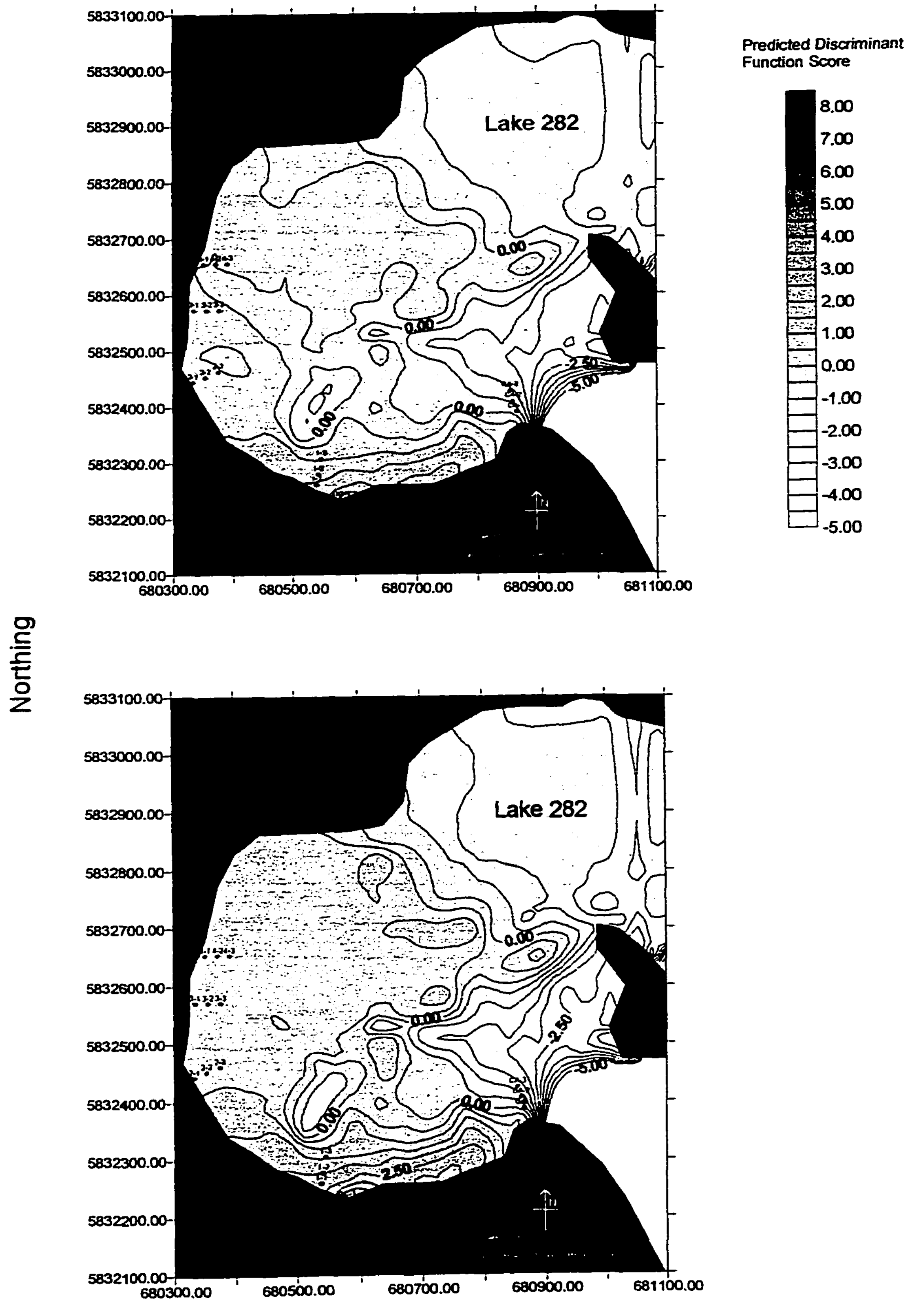
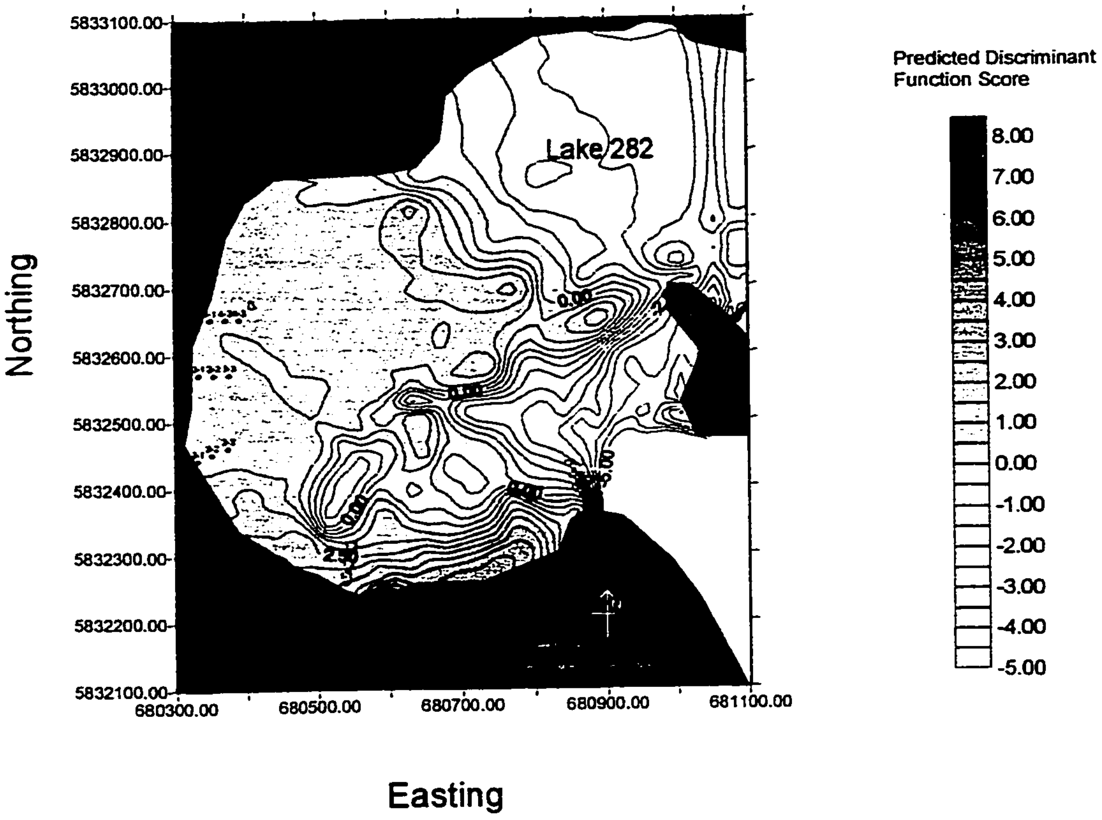


Figure 6.7.3. Plot of Predicted Discriminant Function Scores for 1998 Lake Sediment.



Discussion:

1. *Dispersion of Mine Effluent;*

Musselwhite's effluent plume showed considerable variation in lateral mixing at the 1 m depth between July and August of 1998 due most likely to seasonal differences in Paseminon River flow and patterns of turbulent diffusion. The highest concentrations of effluent in both months were observed to exist along the southern edge of Lake 282's western basin, in a narrow stretch of water that extended from between 50 to 100 meters from shore, and spanned a distance of between 200 m west and 200 m east of the outfall. The relatively high concentrations of mine effluent measured east (upstream) of outfall may be more the result of minor breaches along the discharge channel that were observed during each dye application, rather than the result of dead spot, river back-mixing (Schnoor, 1996). It is possible that effluent escaping the channel could flow along the natural slope of terrain, down through the low lying poor conifer swamp which borders the channel's eastern side to be deposited approximately 150 m away just prior to a slightly raised, rocky peninsula near the relatively narrow river inlet. Such a path could account for effluent concentrations observed east of discharge that were similar to those measured in the vicinity of outfall.

Plots of predicted effluent plumes were based upon the scattered data interpolation of Rhodamine WT dye concentrations collected along transects that attempted to best cover the western basin of Lake 282 in 1998. Grid node density chosen in the generation of these plots, provided a distance between interpolated data points that was slightly less than the distance between nearest transect data points in order to increase the model's accuracy by increasing the likelihood that data point values would be applied directly in the analysis (Keckler, 1997).

Furthermore, due to the relatively small number (<250) and somewhat clustered distribution of observations, the choice was made to use a linear variogram model in order to better assign node interpolation weights, and a quadratic data point search pattern was employed to attempt to give the best two dimensional representation of the data (Keckler, 1997). The spatial prediction of mean effluent concentration produced through this method delineated an area of possible secondary dilution that filled the majority of the lake's western basin, and predicted an area of initial dilution that was perhaps more typical of a shoreline discharge rather than a single point outfall (Dennis et. al., 1994). However, both the monthly and mean effluent predictions were likely accurate given the waste stream's low momentum and potential for easterly seepage caused by water loss through channel breaching, and a lack of measurable ambient current at point of outfall. When combined, these factors could work to limit the rate of initial dilution achieved, and increase the extent of the initial dilution area to include the approximately 400 m long section of southern shoreline predicted by this study.

2. Parameters of Importance;

Aluminum;

Of the common variable data set parameters, total aluminum was identified as a major discriminating variable in Lake 282's sediment during 1997, and at every lake depth in 1998. Mean total aluminum concentrations at the sediment/water interface, and in the sediment of Sample Area 1 were found to be significantly higher than at both the Control Area and Sample Area 4. Concentrations reached a maximum of 0.356 mg/L at the sediment/water interface of Sample Area 1 during 1998, while a mean concentration of 0.263 mg/L (Appendix 3) was also observed to exist within the discharge water for this year. With levels of total aluminum above

0.150 mg/L, at a pH of 7.4, reported as having lethal effects on both goldfish and largemouth bass embryo's and fry (Birge, 1978), this data may suggest a potential water and sediment lake contaminant of concern for Musselwhite's discharge effluent.

The introduction of aluminum to mine effluent may originate during the processing and dissolution of grunerite-garnet-amphibole rich minerals associated with host rock most common to Musselwhite's main ore body (PDC, 1997), while the persistence of aluminum following effluent treatment may stem from an ineffective precipitation of aluminum hydroxide by the single stage INCO SO2/Air treatment process. According to Scott (1987), the solubility of aluminum hydroxide is minimal only between a pH of 5.5 and 6, owing to the amphoteric nature of the metal, and steadily increases logarithmically with deviation in either direction beyond this range. The INCO SO2/Air reactor, although designed to operate between a pH of 6 to 9 (Robbins, 1996), is maintained by Musselwhite at a pH of 8.5 in order to optimize cyanide treatment. It has been suggested by Scott (1987), that wastewater containing aluminum may justify the use of a two-stage metal hydroxide precipitation process, the first at a pH of 6, followed by a second stage at pH 11, with solid-liquid separation between each stage. Such a system would effectively remove aluminum from the slurry solution before other metals, and could be useful to Musselwhite if the slurry treatment process is further suspected of being this metal's source. High aluminum levels in post-treatment effluent may also be influenced by the presence of elevated (>150 mg/L) levels of dissolved sulphate present in the mine's discharge waters. Hem (1968) has reported that sulphate anions will compete with hydroxide for aluminum, and the presence of this competing ligand at concentrations greater than 64mg/L at near neutral pH, can significantly increase the amount of dissolved aluminum in equilibrium with the metal-hydroxide solid. Such competing reactions may interfere with the ability to properly polish this metal from decant waters, and may further contribute to observed effluent levels.

Copper;

Total copper was also identified as a major discriminating variable for 1998 lake sediments. Mean total copper concentrations were found to be significantly higher in the sediments of Sample Areas 1 and 2 when compared to mean values at the Control Area and Sample Area 4. Lake 282 mean concentrations for total copper reached a maximum of 13.63 and 17.39 mg/kg in the sediment of Sample Area 1 for 1997 and 1998 respectively, while relatively higher mean total copper concentrations (30.96 and 45.59 mg/kg) were also observed in sediments collected from the mine's discharge channel during these years. Given that mean sediment total copper values showed a similar increase in concentration over time both near the point of outfall and within the mine's discharge channel, and that levels mostly exceeded published sediment quality guidelines (MOEE, 1992) for lowest effect (Appendix 1) at these sites, copper was identified in this study as a sediment contaminant of potential concern for Lake 282.

Copper may be introduced into Musslewhite's effluent waters either indirectly through ore processing after liberation from the host rock through cyanidation, or directly through the addition of copper sulphate during cyanide destruction. Dissolved copper that remains in the mine's effluent following hydroxide precipitation likely becomes either incorporated into wetland plant tissues through absorption, or is complexed by humic substances upon passing through the mine's wetland. Waters in the Musselwhite area are reported to have a naturally high copper complexing capacity (TVX Gold Inc., 1995), and water draining from Musselwhite's *sphagnum* rich wetland likely contains elevated concentrations of dissolved humic substances in a range of molecular sizes, with the smaller organic acids most likely responsible for the wetland's characteristic "tea" colored water. Rashid (1974) has shown that copper is preferentially sorbed onto dissolved organic substances over other metal cations, and the copper/humic acid complex has been reported to have a metal-humic acid stability second only to that of lead (Bowen, 1966). Sedimentation of copper sorbed humic substances may occur downstream of the wetland through precipitation after binding of these complexes to fine suspended particles of clay minerals, possibly introduced to the effluent from the crushed waste rock lining the discharge channel. According to Forstner and Wittmann (1981), associations between metal-organic species and clay-mineral particulates such as chlorite, a subordinate mineral found in the Musslewhite area (PDC, 1997), can result in a concentration of metals in downstream sedimentary material. Although a significant correlation between total copper and sedimentary organic content (loss upon ignition) was not observed in this study, it remains possible that the output of inorganic particulates washing from the recently constructed discharge channel may exceed what is necessary for organic sedimentation, which would lead to higher inorganic sediment fractions observed in the area of metal deposition.

Zinc;

Total zinc was also identified as a major discriminating variable at each of the surface water, and sediment/water interface lake levels during 1997 and 1998. Mean concentrations for total zinc were found to be significantly higher in both years at Sample Area 1 when compared to those observed at Control and Sample Area 4 at all lake levels. Yearly mean concentrations of total zinc in Lake 282 reached maxima of 0.009 mg/L at the surface, 0.007 mg/L at the sediment/water interface, and 108.69 mg/kg within the sediments of Sample Area 1 in 1998. Mean concentrations of total zinc in the discharge sediments similarly reached a maximum of 87.25 mg/kg in 1998.

While yearly mean concentrations for aqueous total zinc increased only slightly, and remained well below published water quality objectives (MOEE, 1994) both within the receiving lake and discharge channel over time, mean sediment total zinc concentrations at Sample Area 1 increased by 80.05% between 1997 and 1998, and approached the provincial sediment quality guideline of 120 mg/kg in 1998. Discharge sediments similarly showed a yearly mean increase of 113.46% over both years. Due to the possibility of further sediment accumulation of this metal, total zinc was identified as a potential receiving lake contaminant of future concern for Musselwhite's effluent.

Zinc, being a metal that is often preferentially incorporated into the inert lattice possitions of amphiboles (Forstner and Wittmann, 1981), also forms amphoteric metal hydroxide precipitates upon dissolution during processing. According to Stumm and Morgan (1981), the majority of this metal should exist in the insoluble phase at the slurry solution pH found during and after effluent treatment. Low quantities of zinc which remain in equilibrium with the dissolved state, may less preferentially become complexed by wetland introduced humic substances (Rashid 1974), and eventually be precipitated with copper as metal-humic-clay-mineral particulates in the sediment of Lake 282.

Mixing Zone Contaminant Predictions;

Discriminant analysis successfully classified water and sediment quality within the western basin of Lake 282 into three groups based upon combinations of seven parameters; total aluminum, total zinc, total copper, total iron, sulphate, nitrate, and pH. Of the functions generated, the first function was believed to be most indicative of the effluent mixing zone's water and sediment quality for four major discriminating metal species; total aluminum, total copper, total iron, and total zinc, with higher scores generally representing lower quality after appropriate sign transformation. The first lake group, consisting of the three sites situated immediately outside of outfall, was characterized by high scores on the first function in both water and sediment. The second and third groups consisted of the remainder of the sites northwest of outfall and the control sites respectively, and was characterized by relatively lower discriminant scores on the first function in both the water and sediment.

Linear regression equations used to relate mean percent effluent to discriminant functions at individual lake sample sites achieved a high degree of correlation between measured dye concentration and discriminant function score after the natural log transformation of mean percent effluent data. Predictive discriminant score plots, generated from linear regression equations, delineated an area of decreasing water and sediment quality to exist along the southern edge of the lake's western basin, and extend approximately 50m from shore at each of the three levels sampled. Generally, these plots followed a similar pattern for contaminant dispersion as that predicted by the mean percent effluent model for areas of initial and secondary effluent dilution, and were best supported by the actual range of concentrations for total aluminum, copper, and zinc observed between lake sample areas. Predicted discriminant values increased with decreasing distance to outfall, and were found to have greatest increases over time at deeper lake levels. These observations were reflected in similar changes in lake total metal concentrations measured at nearby sampling areas.

Between 1997 and 1998, predictive discriminant plots for the lake's surface showed little change in discriminant contours, possibly due to the relatively minor potential for contaminant accumulation at this depth. The incorporation of total aluminum as a major discriminating variable for Lake 282's surface water during 1998, is however likely indicative of a strong effluent influence for this metal caused by high final effluent concentrations observed during the second year of study (Figure 6.3.1.b)..

At the sediment/water interface, discriminative power in the mixing zone's region of initial dilution increased over both years, and like the lake's surface water, also incorporated total aluminum as a major discriminating variable in the second year. These findings, when taken together, may suggest that the sediment/water interface may be undergoing contaminant accumulation over time, possibly through the combined processes of sedimentation of colloidal complexes removed from the overlying plume (Logan, 1995), and the resuspension of particuates via nearshore wave shear stress (Sly, 1978), since stream shear forces are likely only minor in the area of diminished water quality.

Within Lake 282's sediment, the increase in strength of association between discriminant function score and mean percent effluent between 1997 and 1998 (Table 6.5.1), and the relative increase in contribution of total aluminum and total copper to the first discriminant function in the second year (Tables 6.4.4. a,b) may also be possible indicators of increased contaminant accumulation through the sedimentation of metal sorbed complexes.

Dilution of effluent influenced parameters in Lake 282;

With the exception of total copper in the lake sediment, yearly mean concentrations for all chemical species suspected as being effluent influenced were found to be significantly higher only at Sample Area 1 when compared with other lake areas (Appendix 3). Yearly mean concentrations of sediment total copper were found to be significantly higher at both Sample Area 1 and Sample Area 2 when compared to other lake area concentrations. This would suggest that Lake 282 was capable of diluting all parameters identified as being potentially effluent derived to concentrations approaching those upstream of discharge within a mixing length of 360 m from outfall (Sample Area 3).

Evidence of acid mine drainage;

Despite the finding of elevated concentrations of dissolved sulphate and some total metals in the discharge waters, it is not believed that AMD is an important contributor to final effluent quality at Musselwhite. Mean concentrations for dissolved sulphate of 162 mg/L and 170 mg/L in 1997 and 1998 respectively, were observed in Musselwhite's effluent, and were similar to concentrations measured by the mine, both at point of release from the polishing pond, and at point of release from wetland. Effluent pH showed little observable change from near neutral levels (pH 7.2 -7.3) over time, while concentrations of dissolved metals were found to be mostly low, many being below detection in both the effluent water and at sampling sites on Lake 282 over the length of this study. According to published Musselwhite data, relatively high quality seepage originating from open pit waste rock has been reported as being slightly alkaline (pH >7.5), with a mean sulphate concentration of 247 mg/L, and containing low concentrations of dissolved metals. It may be possible that mine effluent is influenced by such seepage, however it is more likely that observed levels of dissolved sulphate, and elevated concentrations of some total metals are derived more directly from processing.

Cyanide related species in Lake 282;

Evidence of accidental cyanide release or tailings pond seepage into Lake 282 was not found in this study. Weakly acid dissociable cyanide, and total cyanide were only detected in very low concentrations in water samples taken directly from the mine's effluent diversion channel. Thiocyanate, despite being identified as a contaminant of potential concern by the mine, was only detected in trace amounts in the mine's effluent water, and was not detected at all in Lake 282. This data suggests that sufficient dilution, volatilization, or adsorption of dissolved cyanide related species may be quickly achieved within the effluent stream and wetland, and that these contaminants likely do not pose much risk to the aquatic and benthic receiving environments.

Conclusion:

Spatial predictions of contaminant dispersion and deposition for the river mixing zone of Placer Dome Canada's Musselwhite Mine delineated an area of diminished water and sediment quality within and below a 50 m wide, near-shore region that ran approximately 200 m east and west from the mine's outfall. Of the contaminant species that contributed to these predictions, aluminum, copper, and zinc were identified as being possibly of current, or potential environmental concern for the receiving waters or sediments. Of the three depths sampled, the sediment / water interface and lake sediments were suspected of being most influenced by the mine's effluent. Lake 282, within a minimum mixing length of 300 m, was found to be capable of diluting all water and sediment quality parameters carried by the effluent plume to concentrations that were not significantly different from levels found upstream of discharge. Evidence of acid mine drainage, or the release of cyanide related species were not found in this study.

Appendix 1: Data pertaining to the mine's ability to meet Provincial Effluent Limits, Provincial Water Quality Objectives, and Provincial Sediment Quality Guidelines within process effluent water at point of discharge, and within the receiving water body.

A1.1. An Introduction to the Effluent Limits, Water Quality Objectives, and Sediment Quality Guidelines of Ontario.

The Regulation for Metal Mining Effluent Limits established by the Ontario Ministry of Environment and Energy in 1994, outlines process water effluent limits that are applicable to gold mining operations in the province. These effluent limits have been developed through the Municipal and Industrial Strategy for Abatement Program (MISA), and represent levels of contamination achievable by the "best available treatment technology economically achievable". Process effluents are defined by the regulation as effluents that have come into contact with process materials, that may or may not be combined with cooling or storm water, or water originating from cleaning operations. Process effluent produced at the Musselwhite mine is legally required to meet the following criteria; to maintain a pH at all times between 6.0 and 9.5, to be non-toxic to rainbow trout and Daphnia magna, and to meet the maximum daily and monthly average concentrations for total cyanide, lead, copper, nickel, zinc, and arsenic listed in Table 1.1.

Table A1.1.1. Process effluent parameter requirements for Ontario plant discharges.
* (parameters tested for in this study)

Parameter	Maximum Daily Concentration mg/L	Maximum Monthly Average Concentration mg/L
Total Cyanide*	2	1
Total Suspended Solids	30	15
Lead	0.4	0.2
Copper*	0.6	0.3
Nickel*	1	0.5
Zinc*	1	0.5
pH*	> 5.0	> 6.0

The practice of discharging waste water effluent into an aqueous receiving environment is further regulated by the Surface Water Quality Management Policies and the Provincial Water Quality Objectives (PWQO's) outlined in The Ontario Environmental Protection and Water Resources Acts. Provincial Water Quality Objectives represent a minimum satisfactory level of water quality for receiving surface waters, and are designed to be protective of all forms of aquatic life (fish, invertebrates, and plants). A list of PWQO's for parameters investigated in this study follows;

Table A1.1.2.. Provincial Water Quality Objectives for select parameters in Ontario's receiving surface waters.

Parameter:	Conditions of Objective:	PWQO:
Aluminum	pH 6.5-9.0, clay free sample	0.075 mg/L
Ammonia	expressed as un-ionized ammonia (NH_3)*	0.020 mg/L
Copper	none	0.005 mg/L
Cyanide	free cyanide	0.005 mg/L
Iron	none	0.300 mg/L
Nickel	none	0.025 mg/L
pH	none	6.5 to 8.5
Zinc	none	0.030 mg/L

**dependent upon both temperature and pH*

When treated point-source effluents are discharged to a water body, there may exist an area in the receiver contiguous to the point of discharge known as an effluent mixing zone where water quality does not comply with regulating requirements. Mixing zones must be designed to be as small as possible and cannot be used as an alternative to reasonable and practical treatment. Conditions within a mixing zone are not permitted to be acutely lethal to aquatic life, cause irreversible responses to aquatic communities, nor create a barrier to the migration of aquatic life. Also, mixing zones may not impinge upon identified fish spawning areas.

Sediment guidelines exist to protect the aquatic and benthic environments from contaminants that may exist within industrial waste water discharges by setting safe levels for metals, nutrients, and organic compounds. Sediment regulations in Ontario are defined by three levels of effect; the no effect level, lowest effect level, and the severe effect level. No effect is defined as a level at which the chemicals in the sediment will not affect fish or sediment dwelling organisms. The lowest effect level indicates a level of contamination which has no effect on the majority of benthic organisms. At the severe effect level, the sediment is considered to be highly polluted and likely to affect the health of the majority of the benthic species.

Table A1.1.3, A List of Ontario's Provincial Sediment Quality Guidelines for select parameters investigated in this study. (All values expressed in mg/kg (dry weight))

Parameter:	No Effect Level:	Lowest Effect Level:	Severe Effect Level:
Copper	–	16	110
Iron	–	20 000	40 000
Lead	–	31	250
Manganese	–	460	1100
Nickel	–	16	75
Zinc	–	120	820

The following tables contain a summary of data pertaining to the mine's ability to meet provincial effluent limits within its discharge water, provincial water quality objectives within its effluent plume, and sediment quality guidelines below its effluent plume. Similar summaries for Control Area parameters have also been included.

A1.2. Achieved Effluent Requirements

On the dates sampled, mean values for effluent parameters measured in this study met provincial effluent limit requirements for maximum daily allowable limits.

Table A1.2.1. Mean Values for Regulated Parameters in Effluent Water (* represents a value above a provincial limit).

	Effluent Discharge Channel Mean Parameter Values (mg/L) by Month (n=3 sites)											
Regulated Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Copper	No Data		.006	.001	.002	.000	.003	.000	.002	.000	.003	.000
Total Nickel	No Data		.005	.000	BDL		BDL		BDL		BDL	
pH	No Data		7.26	.002	6.69	.50	No Data		7.73	.08	6.97	.13
Total Cyanide	No Data		.011	.001	.028	.001	.012	.000	.006	.000	.034	.002
Total Zinc	No Data		.004	.001	.001	.000	.001	.000	.002	.000	.002	.000

A1.3. Parameters in Non-Compliance with Provincial Water Quality Objectives and Provincial Sediment Quality Guidelines at Sample Area 1.

Within the first sampling area of the receiving water body, mean surface values for both total aluminum and total copper exceeded provincial water quality objectives with frequencies of 4 out of 6, and 2 out of 6 months respectively. At the sediment/water interface, mean total aluminum concentrations exceeded PWQO's in every month sampled, while unionized ammonia exceeded its PWQO in 3 of 6 months. At the sediment level, total copper exceeded the lowest effect level with a frequency of 2 out of 6 months, while total iron and total nickel exceeded lowest effect in every month sampled. Mean values for total manganese in the sediment exceeded the severe effect level in every month sampled.

Table A1.3.1. Mean Provincial Water Quality Objective Related Parameter Values for Sample Area 1 Surface Waters (values marked with an * lie above a provincial objective)

	Sample Area 1, Surface Water Mean Parameter Values (mg/L) by Month (n=3)											
PWQO Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Aluminum	.143*	.022	.080*	.005	.096*	.003	.067	.002	.076*	.000	.069	.002
Ammonia	.035	.002	.078	.046	.053	.000	BDL		BDL		.072	.013
Total Copper	.008*	.002	.004	.001	.004	.000	.005	.001	.005	.001	.011*	.006
Total Cyanide	BDL		BDL		BDL		BDL		BDL		BDL	
Total Iron	.166	.022	.090	.007	.109	.018	.106	.009	.088	.014	.086	.003
Total Nickel	.011	.007	.005		.008	.003	.056	.030	.018		.009	
pH	7.38	.03	7.39	.06	7.24	.05	7.15	.03	7.52	.12	6.76	.12
Total Zinc	.009	.001	.011	.001	.005	.003	.007	.003	.009	.005	.010	.003

Table A1.3.2. Mean Provincial Water Quality Objective Related Parameter Values for Sample Area 1 Sediment / Water Interface (values marked with an * lie above provincial objectives)

	Sample Area 1, Sediment / Water Interface Mean Parameter Values (mg/L) by Month (n=3)											
PWQO Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Aluminum	.177*	.04	.258*	.149	.164*	.002	.257*	.061	.279*	.012	.492*	.035
Ammonia	.025*		.069*	.018	BDL		BDL		BDL		.098*	
Total Copper	.008	.002	.005	.001	.004	.002	.005	.001	.007	.001	.012	.000
Total Cyanide	BDL		BDL		BDL		BDL		BDL		BDL	
Total Iron	.153	.009	.123	.014	.170	.021	.138	.007	.301	.083	.143	.037
Total Nickel	BDL		BDL		.005	.001	.020	.004	BDL		BDL	
pH	7.38	.02	7.41	.03	7.25	.08	7.25	.05	7.38	.04	6.74	.07
Total Zinc	.006	.001	.004	.001	.003	.000	.005	.001	.007	.001	.009	.001

Table A1.3.3. Monthly Mean Provincial Sediment Quality Guideline Related Parameter Values for Sample Area 1 (values marked with an * lie above the lowest effect level, ** represents values above severe effect level)

	Sample Area 1, Sediment Mean Parameter Values (mg/kg) by Month (n=3)											
PSQG Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Copper	14.1	1.2	11.4	1.7	15.3	3.2	18.7*	1.7	16.4*	1.7	16.9*	1.9
Total Iron	26948 *	3363	17313 *	2262	22390 *	2441	31233 *	6686	37300 *	4931	32400 *	4562
Total Manganese	2974 **	561	1810 **	126	2504 **	603	4166 **	1470	3846 **	99.5	3586 **	1035
Total Nickel	28.5*	4.5	19.2*	3.3	23.9*	3.8	43.1*	6.8	41.6*	4.5	35.6*	5.5
Total Zinc	74.7	10.8	47.9	7.8	58.5	8.0	106.8	17.9	107.7	14.9	111.5	12.6

A1.4. Parameters in Non-Compliance with Provincial Water Quality Objectives and Provincial Sediment Quality Guidelines at Sample Area 4.

Within Sample Area 4, mean values for total aluminum, total copper, and unionized ammonia exceeded provincial water quality objectives with frequencies of 3 out of 6, 2 out of 6, and 2 out of 6 months respectively. At the sediment/water interface, mean total aluminum concentrations exceeded PWQO's in every month sampled, while both unionized ammonia and total copper exceeded PWQO's in 3 of 6 months sampled. At the sediment level, total iron exceeded lowest effect PSQG in two out of six months, while mean total manganese concentrations exceed the lowest effect level in 5 of 6 months sampled, and exceeded the severe effect level on two of those five occasions.

Table A1.4.1. Mean Provincial Water Quality Objective Related Parameter Values for Sample Area 4 Surface Waters (values marked with an * lie above a provincial objective)

	Sample Area 4, Surface Water Mean Parameter Values (mg/L) by Month (n=3)											
PWQO Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Aluminum	.122*	.013	.075	.007	.089*	.024	.041	.010	.088*	.008	.046	.005
Ammonia			.142*	.016							.079*	.019
Total Copper	.0069 *	.0011	.0019	.0004	.0022	.0007	.0034	.0008	.0051 *	.0002	.0033	.0008
Total Cyanide												
Total Iron	.147	.012	.088	.007	.124	.013	.119	.014	.172	.058	.070	.004
Total Nickel							.007	.001				
pH	7.35	.01	7.45	.03	7.26	.11	7.64	.06	7.48	.04	7.06	.04
Total Zinc	.005	.001	.002	.001	.001	.001	.002	.001	.004	.001	.003	.001

Table A1.4.2. Mean Provincial Water Quality Objective Related Parameter Values for Sample Area 4 Sediment / Water Interface (values marked with an * lie above provincial objectives)

	Sample Area 4, Sediment / Water Interface Mean Parameter Values (mg/L) by Month (n=3)											
PWQO Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Aluminum	.147*	.018	.094*	.024	.129*	.008	.096*	.024	.115*	.014	.104*	.025
Ammonia	.025*		.083*	.041	.012	.011					.085*	.013
Total Copper	.006*	.001	.003	.001	.005*	.001	.010*	.001	.006*	.001	.005	.001
Total Cyanide												
Total Iron	.174	.019	.113	.030	.169	.014	.145	.013	.147	.014	.138	.028
Total Nickel							.009	.001			.006	.001
pH	7.44	.02	7.50	.01	7.58	.05	762	.04	7.59	.04	7.15	.03
Total Zinc	.004	.001	.002	.001	.002	.001	.004	.001	.006	.001	.004	.001

Table A1.4.3. Monthly Mean Provincial Sediment Quality Guideline Related Parameter Values for Sample Area 4 (values marked with an * lie above the lowest effect level, ** represents values above severe effect level)

	Sample Area 4, Sediment Mean Parameter Values (mg/kg) by Month (n=3)											
PSQG Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Copper	10.2	1.9	7.6	1.6	11.9	.2	15.2	2.5	12.3	.145	10.7	1.15
Total Iron	9066	1401	7017	1296	16253	374	23366 *	4470	26433 *	4731	17800	2406
Total Manganese	509*	134	382	95	823*	47	1477 **	408	1318 **	310	957*	191
Total Nickel	11.9	1.8	9.3	1.9	20.2*	0.5	27.0*	5.1	33.9*	6.9	21.1	3.1
Total Zinc	30.1	6.2	24.4	5.9	39.1	10.9	74.3	14.4	69.3	9.1	62.3	10.1

A1.5. Parameters in Non-Compliance with Provincial Water Quality Objectives and Provincial Sediment Quality Guidelines at Control Area.

Within the surface waters of the Control Area mean total aluminum concentrations exceeded its PWQO on three occasions. Mean unionized ammonia and total copper concentrations both exceeded PWQO's on 2 of the 6 months sampled. At the sediment/water interface, mean total aluminum concentrations exceeded PWQO on every occasion sampled, while both unionized ammonia and total copper concentrations exceeded PWQO's with frequencies of 2 out of 6 and 1 out of 6 months respectively. Within the sediment, mean concentrations of total iron, and total nickel exceeded the lowest effect level for PSQG on 3 of 6 and 5 of 6 occasions, while total manganese exceed the severe effect level on every ocassion sampled.

Table A1.5.1. Monthly Mean Provincial Water Quality Objective Related Parameter Values for Control Area Surface Waters (values marked with an * lie above a provincial objective)

	Control Area, Surface Water Mean Parameter Values (mg/L) by Month (n=3)											
PWQO Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Aluminum	.099*	.002	.068	.026	.101*	.003	.051	.006	.086*	.006	.062	.006
Ammonia	.020		.132*	.008	BDL		BDL		BDL		.098*	
Total Copper	.007*	.001	.003	.001	.002	.001	.005	.001	.006*	.001	.004	.001
Total Cyanide	BDL		BDL		BDL		BDL		BDL		BDL	
Total Iron	.146	.019	.328	.025	.128	.006	.123	.005	.119	.006	.082	.005
Total Nickel							.009	.003				
pH	7.37	.03	7.53	.01	7.51	.01	7.63	.09	7.71	.06	7.01	.09
Total Zinc	.005	.001	.002	.001	.001	.001	.004	.001	.004	.001	.003	.001

Table A1.5.2. Monthly Mean Provincial Water Quality Objective Related Parameter Values for Control Area Sediment / Water Interface (values marked with an * lie above provincial objectives)

	Control Area, Sediment / Water Interface Mean Parameter Values (mg/L) by Month (n=3)											
PWQO Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Aluminum	.147*	.018	.094*	.024	.129*	.008	.096*	.024	.115*	.014	.104*	.025
Ammonia	.020		.083*	.041	.012	.013					.085*	.013
Total Copper	.005	.001	.003	.001	.005	.001	.010*	.001	.005	.001	.005	.001
Total Cyanide	BDL		BDL		BDL		BDL		BDL		BDL	
Total Iron	.174	.019	.113	.030	.169	.014	.145	.013	.147	.014	.132	.028
Total Nickel	BDL		BDL		BDL		.009		BDL		.006	
pH	7.44	.02	7.50	.01	7.58	.05	7.62	.04	7.59	.04	7.15	.03
Total Zinc	.004	.001	.002	.001	.002	.001	.004	.001	.006	.001	.004	.001

Table A1.5.3. Mean Provincial Sediment Quality Guideline Related Parameter Values for Control Area (values marked with an * lie above the lowest effect level, ** represents values above severe effect level)

	Control Area, Sediment Mean Parameter Values (mg/kg) by Month (n=3)											
PSQG Parameter	July 1997	SE of mean	Aug 1997	SE of mean	Sept 1997	SE of mean	June 1998	SE of mean	July 1998	SE of mean	Aug 1998	SE of mean
Total Copper	8.0	1.0	7.5	.8	9.5	1.0	11.3	2.3	11.7	.7	11.8	.5
Total Iron	15494	2655	14713	1356	17695	1617	34400 *	7076	39700 *	1686	36566 *	1117
Total Manganese	1902 **	385	1999 **	287	2431 **	256	4013 **	1408	4753 **	267	3840 **	105
Total Nickel	22 *	1.1	14.9	1.8	18.4 *	2.4	34.8 *	8.9	41.3 *	1.9	40*	1.3
Total Zinc	42.9	10.2	30.9	.27	34.3	2.2	66.9	13.5	78.1	.34	82.8	1.6

Table A1.6. Monthly frequencies for parameters in noncompliance with Provincial Water Quality Objectives or Provincial Sediment Quality Guidelines.

		Sample Area 1		Sample Area 4		Control Area	
Parameter:	Lake Level:	1997	1998	1997	1998	1997	1998
Total Aluminum	Surface	1.00	0.33	0.67	0.33	0.67	0.33
Total Aluminum	Sed / Water Interface	1.00	1.00	1.00	1.00	0.67	0.33
Total Copper	Surface	0.33	0.33	0.33	0.33	0.33	0.33
Total Copper	Sed / Water Interface	0.00	0.00	0.67	0.67	0.33	0.33
Total Copper	Sediment	0.00	1.00	0.00	0.00	0.00	0.00
Ammonia	Surface	0.00	0.00	0.33	0.33	0.33	0.33
Ammonia	Interface	0.67	0.33	0.67	0.33	0.33	0.33
Total Iron	Sediment	1.00	1.00	0.00	0.67	0.00	1.00
Total Manganese	Sediment	1.00	1.00	0.33	1.00	1.00	1.00
Total Nickel	Sediment	1.00	1.00	0.33	0.67	0.67	1.00

Table A1.7. Yearly mean values (n=9) for parameters in noncompliance with provincial objectives or guidelines expressed as a percentage of objective or guideline. * represents % of lowest effect level, ** represents % of severe effect level. (Error term representative of standard error in mean estimate).

	Sample Area 1		Sample Area 4		Control Area	
Parameter	1997 % of objective or guideline	1998 % of objective or guideline	1997 % of objective or guideline	1998 % of objective or guideline	1997 % of objective or guideline	1998 % of objective or guideline
Total Aluminum (surface)	142.67% +/-16.00%	94.67% +/-2.66%	126.67% +/-14.67%	77.33% +/-10.67%	118.67% +/-12.00%	88.00% +/-0.08%
Total Copper (surface)	100.00% +/- 20.00%	140.00% +/- 28.5%	80.00% +/- 25.00%	80.00% +/- 25.00%	80.00% +/- 25.00%	100.00% +/- 20.00%
Ammonia (surface)						
Total Aluminum (interface)	265.33% +/-62.67%	457.33% +/-148.00%	144.00% +/-16.00%	204.00% +/-34.66%	164.00% +/-16.00%	140.00% +/-14.67%
Total Copper (interface)	120.00% +/-20.00%	160.00% +/-20.00%	60.00% +/-20.00%	100.00% +/-20.00%	100.00% +/-20.00%	140.00% +/-20.00%
Ammonia (interface)						
Total Copper (sediment)	*85.19% +/-7.75%	*108.68% +/-6.02%	*62.00% +/-5.94%	*79.65% +/-6.46%	*51.88% +/-4.70%	*72.36% +/-5.12%
Total Iron (sediment)	*111.09% +/-5.135%	*168.22% +/-14.45%	*53.89% +/-7.54%	*112.67% +/-9.96%	*79.84% +/-5.38%	*184.44% +/-11.29%
Total Manganese (sediment)	**220.90% +/-26.73%	**351.45% +/-54.53%	**51.90% +/-7.45%	**113.73% +/-15.90%	**191.91% +/-16.09%	**382.01% +/-39.82%
Total Nickel (sediment)	*149.19% +/-14.13%	*250.69% +/-19.14%	*86.18% +/-11.42%	*170.90% +/-20.20%	*115.18% +/-8.59%	*241.80% +/-17.90%

Appendix 2: Locations of Sampling Areas and Sampling Sites used on Lake 282.

Table A.2.1. Sampling area and site locations on Lake 282 as measured from effluent outfall.

Sample Area / Site	Distance Downstream from Outfall (m)	Distance from Shore (m)	Orientation to First Site in Area, Transect Heading (Degrees)	GPS Easting	GPS Northing
1/1	40	2	10		
1/2	40	22	10		
1/3	40	42	10		
2/1	360	2	76	680319.6	5832437.4
2/2	360	22	76		
2/3	360	42	76		
3/1	480	2	107	680347.8	5832500.7
3/2	480	22	107		
3/3	480	42	107		
4/1	560	2	112	680328.0	5832681.8
4/2	560	22	112		
4/3	560	42	112		
Control/1	-100	2	110		
Control/2	-100	22	110		
Control/3	-100	42	110		

Appendix 3. Possible parameters under effluent influence in Lake 282

Parameters identified by LSD testing as being significantly higher at Sample Area 1 when compared with concentrations at Sample Area 4 and at Control (Table A3.1.), were classified as being possibly effluent influenced in this study.

Table A3.1. Summary of parameters identified as being significantly ($p<0.05$) different in concentration at Sample Area 1 than at both Sample Area 4 and at Control Area through LSD testing.

	Year:	
Data Set:	1997	1998
Surface Water	Total Zinc (higher)	Total Zinc (higher)
Sediment/Water Interface	Total Aluminum (higher) Total Zinc (higher)	Total Aluminum (higher) Total Zinc (higher) pH (lower)
Sediment	Extractable Nickel (higher) Total Aluminum (higher) Total Copper (higher) Total Iron (higher) Total Nickel (higher) Total Zinc (higher) paste pH (higher)	Total Aluminum (higher) Total Copper (higher) Total Zinc (higher)

Of the parameters identified in Table A3.1., none were found to be significantly different in concentration at Sample Area 4 when compared to Control Area mean yearly values (Table A3.2.).

Table A3.2. Summary of parameters identified as being significantly ($p<0.05$) different in mean value at Sample Area 4 than at Control Area through LSD testing.

	Year:	
Data Set:	1997	1998
Surface Water	pH (higher)	
Sediment/Water Interface	pH (lower)	Total Iron (higher)
Sediment	Ammonia (higher)	Extractable Nickel (higher) Loss Upon Ignition (higher) Paste pH (lower)

Appendix 4: 1997 preliminary plume study data.

Figure A4.1 a, b. Tracer dye concentrations as measured along transects connecting sampling sites belonging to Sample Area 1 during 1997 effluent plume studies (depth = 1m). Sample Area and Sample Site positions are represented by a dashed line, and are labeled numerically according to area and site number. TIP; transect initiation point.

Figure A4.1.a. Rhodamine WT concentration in July 1997 effluent-tracer plume as measured along a transect connecting sampling sites belonging to Sample Area 1.

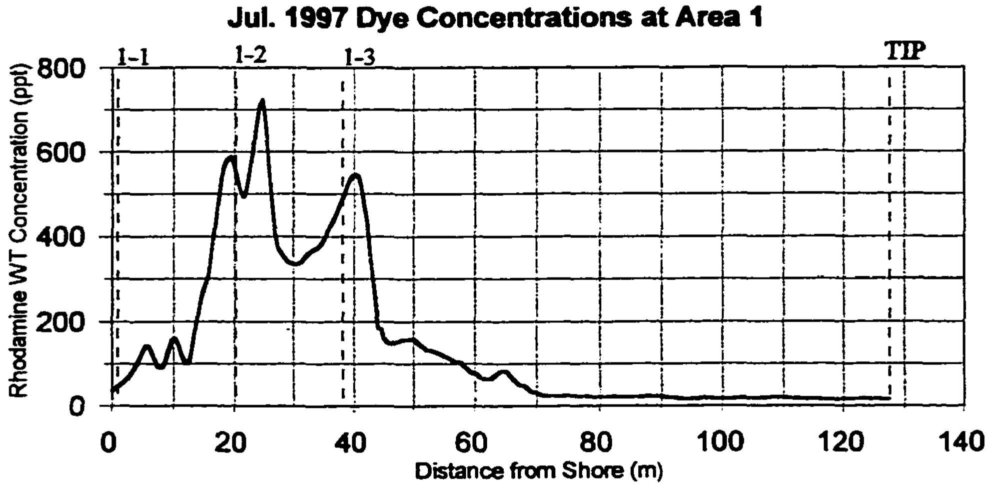


Figure A4.1.b. Rhodamine WT concentration in August 1997 effluent-tracer plume as measured along a transect connecting sampling sites belonging to Sample Area 1.

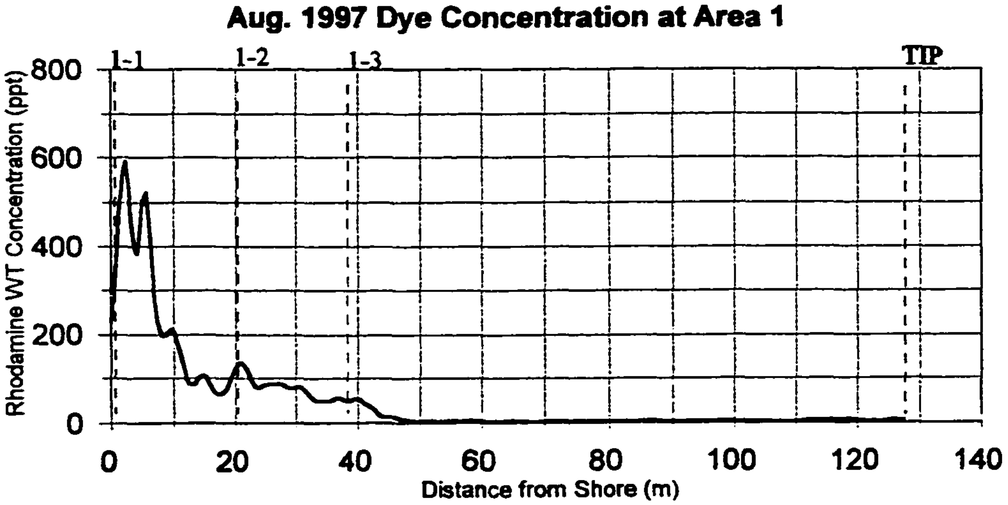


Figure A4.2 a, b. Tracer dye concentrations as measured along transects connecting sampling sites belonging to Sample Area 2 during 1997 effluent plume studies (depth = 1m). Sample Area and Sample Site positions are represented by a dashed line, and are labeled numerically according to area and site number. TIP; transect initiation point.

Figure A4.2.a. Rhodamine WT concentration in July 1997 effluent-tracer plume as measured along a transect connecting sampling sites belonging to Sample Area 2.

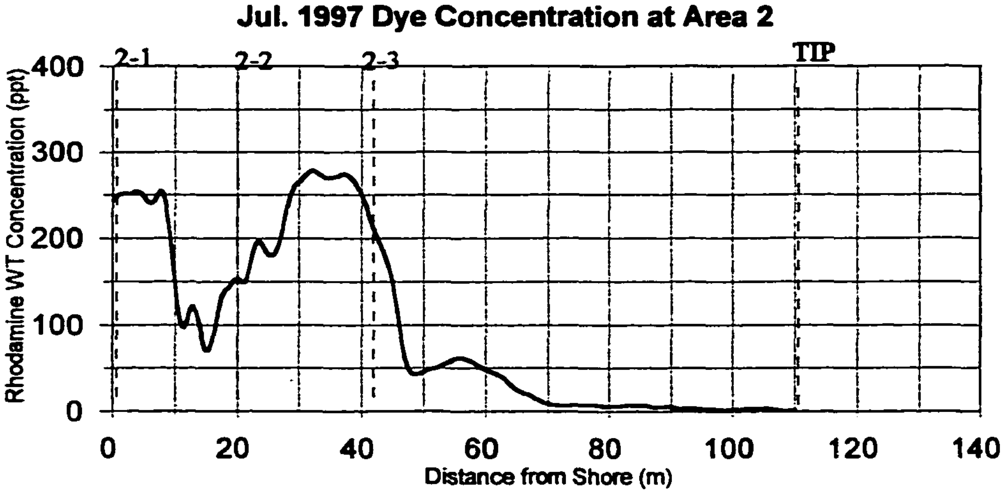


Figure A4.2.b. Rhodamine WT concentration in August 1997 effluent-tracer plume as measured along a transect connecting sampling sites belonging to Sample Area 2.

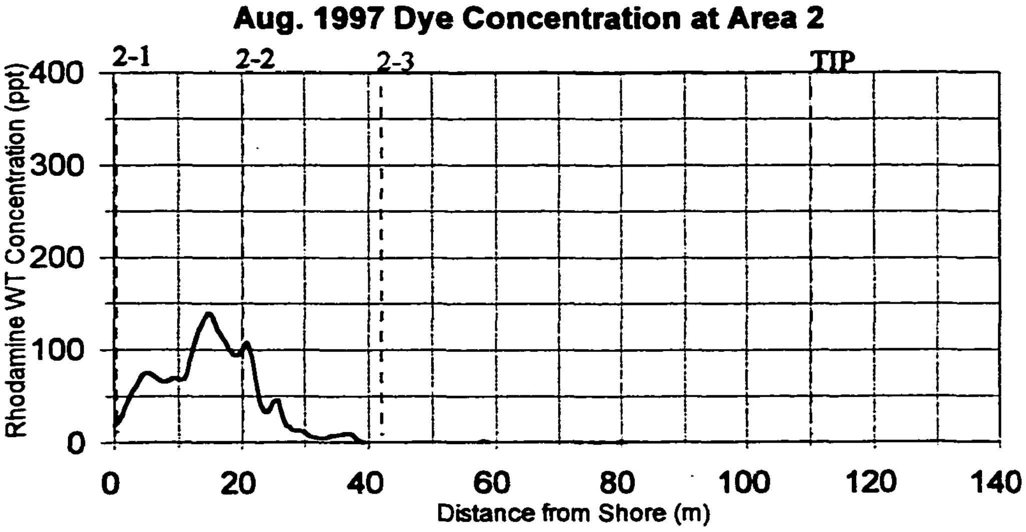


Figure A4.3 a, b. Tracer dye concentrations as measured along transects connecting sampling sites belonging to Sample Area 3 during 1997 effluent plume studies (depth = 1m). Sample Area and Sample Site positions are represented by a dashed line, and are labeled numerically according to area and site number. TIP; transect initiation point.

Figure A4.3.a. Rhodamine WT concentration in July 1997 effluent-tracer plume as measured along a transect connecting sampling sites belonging to Sample Area 3.

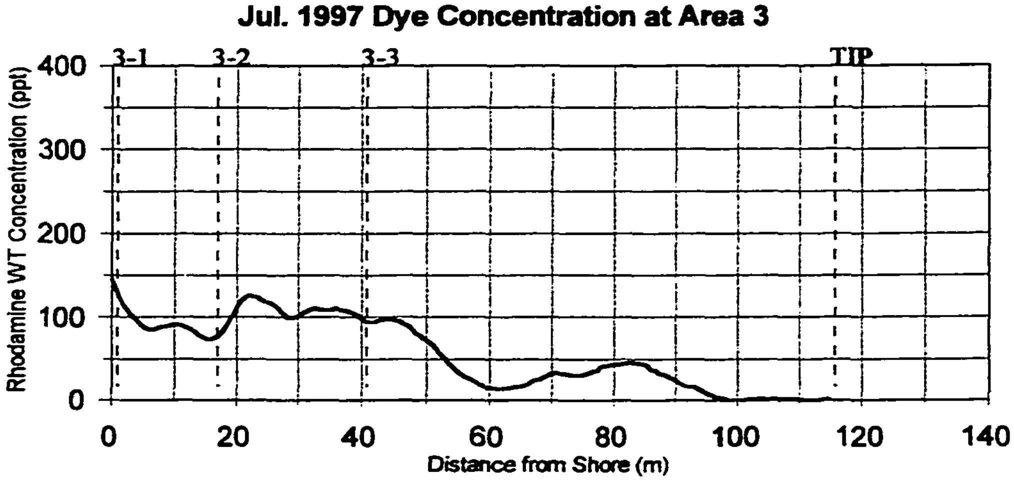


Figure A4.3.b. Rhodamine WT concentration in August 1997 effluent-tracer plume as measured along a transect connecting sampling sites belonging to Sample Area 3.

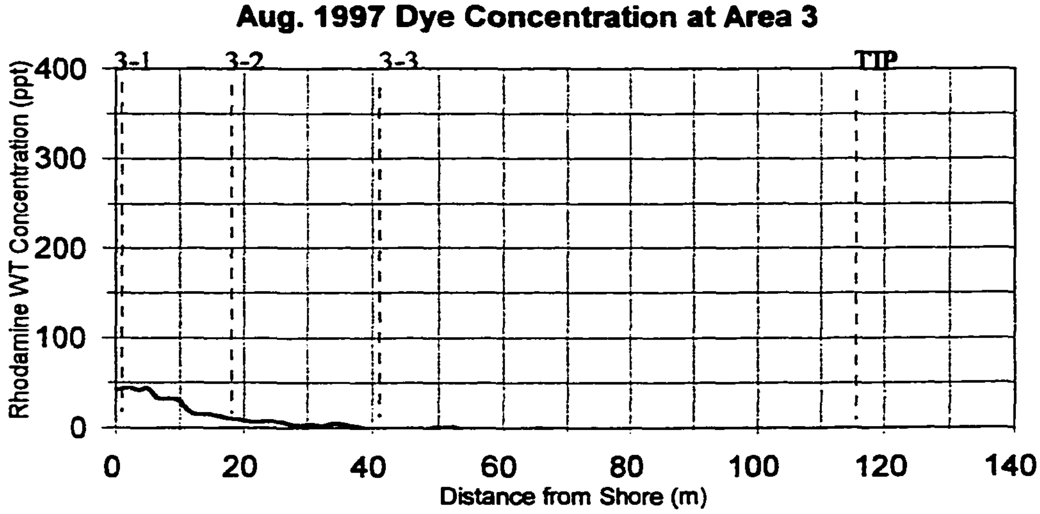


Figure A4.4 a, b. Tracer dye concentrations as measured along transects connecting sampling sites belonging to Sample Area 4 during 1997 effluent plume studies (depth = 1m). Sample Area and Sample Site positions are represented by a dashed line, and are labeled numerically according to area and site number. TIP; transect initiation point.

Figure A4.4.a. Rhodamine WT concentration in July 1997 effluent-tracer plume as measured along a transect connecting sampling sites belonging to Sample Area 4.

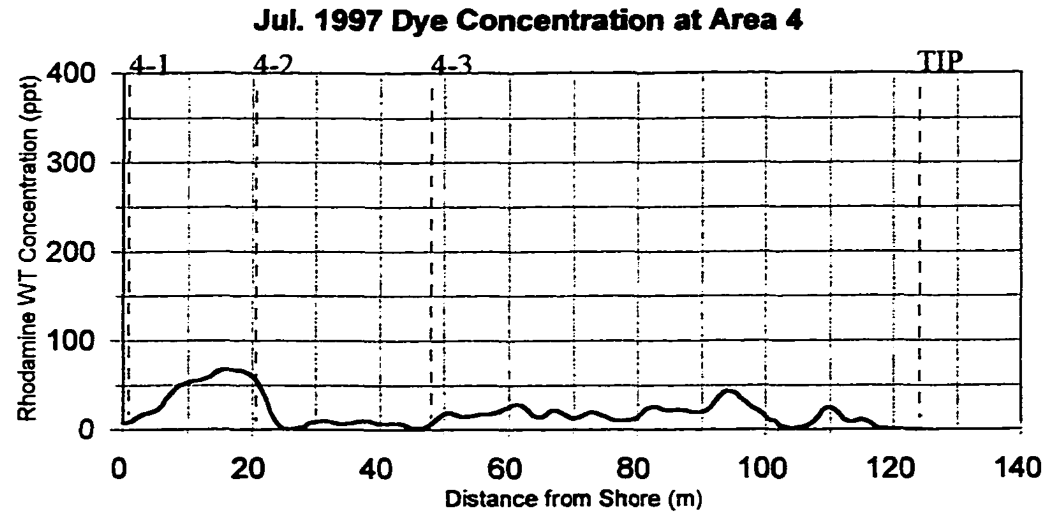
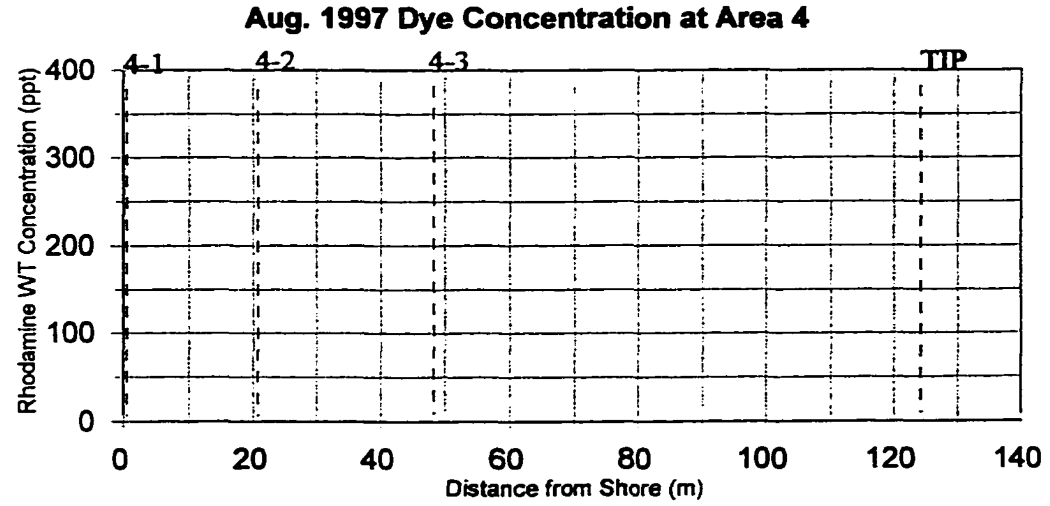


Figure A4.4.b. Rhodamine WT concentration in August 1997 effluent-tracer plume as measured along a transect connecting sampling sites belonging to Sample Area 4.



Literature Cited

American Soil and Plant Analysis Coucil, 1992. *Handbook on reference methods for soil analysis.* Georgia University Station, GA, pp. 133-138.

Birge, W. J., 1978. Aquatic toxicology of trace elements of coal and fly ash. Energy and Environmental Stress in Aquatic Systems. J.H. Thorp and J.W. Gibbbons (eds.). U.S. Dept. Energy, Tech. Info. Centre. pp. 219-239.

Brusseau., M.L., C.P. Gerba, I.L. Pepper, 1996. *Pollution Science.* Academic Press, New York. pp. 199-209.

Bowen, H.J.M., 1966. Residence times of heavy metals in the environment. Proc. Symp. Heavy. Met. Environ., Toronto 1975, I/1-9.

Burris, J.E., 1984. *Treatment of Mine Drainage by Wetlands*, Pennsylvania State University, University Park, PA.

Carter, H.H., 1974. The measurement of rhodamine tracers in natural systems by fluorescence. Rapports et proces-verbaux des reuvious. couseil international pour l'exploration de lamer, v.167, pp. 193-200.

Cressie, N.A.C., 1991. *Statistics for Spatial Data*, John Wiley and Sons Inc., New York, p 900.

Dennis, P., G. Rutherford, D. Spry, 1994. Deriving receiving water based point-source effluent requirements for Ontario waters. Ministry of Environment and Energy.

Evangelou, V.P., Y.L. Zang, 1995. Pyrite oxidation mechanisms and acid mine drainage prevention. Critical Reviews in Environmental Science and Technology, v.25, pp. 141-199.

Falbo M.B., and Weaks. T.E., 1990. A comparison of Eichhomia crassipes (Pontederiaceae) and Sphagnum quincuefarium (Sphagnaceae) in Treatment of Acid Mine Water. Economic Botony, v. 44, pp. 40-49.

Feasby, D.G., 1993. The mine environment neutral drainage (MEND) program, CIM Bulletin, v 86, p 971.

Ferguson, K.D., P.M. Erikson, 1987. An overview of methods to predict acid mine drainage - Proceedings Acid Mine Drainage / Workshop March 23-26. Conservation and Protection Environment Canada. pp 215-244.

Forstner, U., G.T.W. Wittmann, 1981. *Metal Pollution in the Aquatic Environment.* Springer-Verlag, Berlin Heidelberg, pp 220-227.

Gormely, L., T.W. Higgs, R.U. Kistristz, A. Sobolewski, 1990. Assessment of wetlands for gold mill effluent treatment. Gormely Process Engineering, Vancouver, BC. 63pp.

Gould, W.D., D. Bechard, L. Lortie, 1994. The nature and role of microorganisms in the tailings environment. In: Short course Handbook on environmental geochemistry of sulfide mine wastes. J.L. Jambor and D.W. Blowes (eds.). Mineralogical Association of Canada. Pp.186-199.

Gowda, T.P., 1980. Stream tube model for water quality prediction in mixing zones of shallow rivers. Ontario Ministry of the Environment.

Gowda, T.P., 1984. Critical point method for mixing zones in rivers. Journal of Environmental Engineering. v 110, pp.244-261.

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

Harris, A.G., S.C. McMurray, P.W.C. Uhlig, J.K. Jeglum, R.F. Foster, G.D. Racey, 1996, *Field Guide to the Wetland Ecosystem Classification for Northwestern Ontario*. Ontario Ministry of Natural Resources NWST Field Guide: FG-01. Lehto Printers Ltd. Thunder Bay, Ontario. p. 48.

Hem, J.D., 1968. Study and interpretation of the chemical characteristics of natural waters. U.S. Geological Survey Water Supply Paper 1473.

Herbert, D.W.M., D.S. Shurben, 1965. The susceptibility of salmonid fish to poisons under estuarine conditions II. Ammonium chloride. International Journal of Air and Water Pollution, v9, pp.89-91.

Higgs, T., L. Gormely, K. Liber, A. Sobolewski, A. Maynard, B. Szczachor, 1992. *Technical guide for the environmental management of cyanide in mining*. T.W. Higgs Associates Ltd. British Columbia Technical and Research Committee on Reclamation.

Howard, P.H., 1991. *Handbook of environmental degradation rates*. Lewis Publishers inc., Chelsea, Michigan.

Kilpatric, F.A., 1970. Dosage requirements for slug injections of rhodamine BA and WT dyes. U.S. Geological Survey Research Professional Paper 700-B, Department of the Interior, pp.250-253.

Kilpatric, F.A., E.D. Cobb, 1985. Measurement of discharge using tracers. U.S. Geological Survey Techniques of Water-Resources Investigations, Department of the Interior, Book 3, Ch. A16.

Keckler, D., 1997. *Surfer for Windows*, Golden Software, Inc. CO, pp.90-155.

Lee, P.F. and Stewart, J.M., 1981. Ecological relationships of wild rice, <u>*Zizania aquatica*</u> L. I. A model for among-site growth. Canadian Journal of Botany, v 59, pp. 2140-2151.

Logan, T. J., 1995. Water Quality. Environmental Hydrology. W.J. Elliot, A.D. Ward (eds.) Lewis Publishers, New York, pp.321-333.

Meeussen, J.C.L., E.J.M. Temminghoff, M.G. Keizer, I. Novozamsky, 1899. Spectrophotometric determination of total cyanide, iron cyanide complexes, free cyanide, and thiocyanate in water by a continuous flow through system. Department of soil science and plant nutrition, Agricultural University Wageningen, The Analyst, v114.

MOEE, 1992. Guidelines for the protection and management of aquatic sediment quality in Ontario, Water Resources Branch of the Ontario Ministry of the Environment.

MOEE, 1994. Water management policies, guidelines and provincial water quality objectives of the Ministry of Environment and Energy.

Placer Dome Canada Ltd., 1997. Musselwhite mine geology summary.

Personal Communication, Steve Parsons, Musselwhite Mill Supervisor.

Rashid, M.A., 1974. Adsorption of metals on sedimentary and peat humic acids. Chemical Geology, v.13, pp.115-123.

Ritchie, A.I.M., 1994. The Waste-rock Environment. In: Short course Handbook on environmental geochemistry of sulfide mine wastes. J.L. Jambor and D.W. Blowes (eds.). Mineralogical Association of Canada. pp. 133-145.

Robbins, G.H., 1996. Historical development of the INCO SO2/AIR cyanide destruction process. CIM Bulletin, v89, p.63-69.

Sly, P.G., 1986. Sediments and Water Interactions. Springer-Verlag New York Inc. New York. pp 3-63.

Smith, A.C.S., T.I. Mudder, 1991. *Chemistry and Treatment of Cyanidation Wastes.* Mining Journal Press, London, UK.

Scott, J.S., 1987. *Mine and Mill Wastewater Treatment.* Environment Canada Report EPS 2/MM/3.

Schnoor, J.L., 1996. *Environmental Modelling.* J. Wiley Press.

Searle, P.L., 1984. The Berthelot or indophenol reaction and its use in the analysis chemistry of nitrogen, The Analyst, v109, pp.549-565.

Stumm, W., J.J. Morgan, 1981. *Aquatic Chemistry*, Second Edition, John Wiley and Sons, New York, N.Y..

TVX Gold Inc. and Placer Dome Canada Ltd., 1995. Musselwhite project environmental impact statement and comprehensive study report. Environmental Description, v2.

Ward, A.D., and Elliot. W.J., 1995. *Environmental Hydrology*. Lewis Publishers Inc., NY, New York. pp. 311-335.

Wetzel, R. G., 1983. *Limnology*. Saunders College Publishing, New York, N.Y., p. 223-253.

Wielder, K.R., 1990. Metal cation binding to *sphagnum* peat and sawdust: relation to wetland treatment of metal polluted waters. Water, Air, and Soil Pollution, v 53, pp. 391-400.

Wright, T., 1993. *Skalar automated cyanide analysis*. Skalar B.V., Breda, Denmark, pp 9-48.

Vitt, D.H., and N.G. Slack. 1975. An analysis of the vegetation of *Sphagnum*-dominated kettlehole bogs in relation to environmental gradients. Canadian Journal of Botany. v.53, pp 322-359.