5-1-1992

ANTHROPOGENIC EFFECTS ON NEW HAMPSHIRE SURFACE WATER QUALITY: LONG TERM EVIDENCE FROM LAKE SEDIMENTS

Henri E. Gaudette
University of New Hampshire

W. Berry Lyons
University of New Hampshire

Recommended Citation
Gaudette, Henri E. and Lyons, W. Berry, "ANTHROPOGENIC EFFECTS ON NEW HAMPSHIRE SURFACE WATER QUALITY: LONG TERM EVIDENCE FROM LAKE SEDIMENTS" (1992). *NH Water Resources Research Center Scholarship*. 135.
https://scholars.unh.edu/nh_wrrc_scholarship/135

ANTHROPOGENIC EFFECTS ON NEW HAMPSHIRE SURFACE WATER QUALITY: LONG TERM EVIDENCE FROM LAKE SEDIMENTS

Henri E. Gaudette
W. Berry Lyons
Earth, Ocean, Space Science Center

Technical Completion Report #56

FY 90 Grant 14-08-0001-G1576

Water Resource Research Center
University of New Hampshire
Durham, New Hampshire

May 1992

The research on which this report is based was financed in part by the United States Department of the Interior (USGS) as authorized by the Water Resources Research Act of 1984 (PL 98-242), the New Hampshire Water Resource Research Center, and the University of New Hampshire.

TABLE OF CONTENTS

I. INTRODUCTION:

Since the mid 1970's considerable attention has been focused on surface water and groundwater contamination near hazardous waste disposal sites in which large volumes of toxic and/or carcinogenic industrial and chemical wastes were disposed. This focus has brought about much needed Federal and State regulations regarding the generation, transportation, storage and disposal of hazardous wastes. But it has also developed an inattention and lack of concern to the contamination threat due to municipal or sanitary landfills in which household, municipal and low volumes of industrial waste are disposed, and which outnumber hazardous waste landfills by 20 to 1 (Wentz, 1989; U.S. EPA, 1984; and U.S. EPA Office of Water-Waste Management, 1980).

It is estimated that every household contributes a gallon or more of hazardous waste per year to sanitary landfills, therefore hundreds of millions of gallons of the same chemicals involved under the Resource Conservation and Recovery Act of 1976 and the Superfund Act of 1980 are being sent to the local town landfill (Wentz, 1989; Lee, Jones and Ray, 1986). In 1981 there were approximately 1,350 operating sanitary landfills in New England. In 1983 that number was down to 950 (van der Leeden, Troise and Todd, 1990). At that time the United States Environmental Protection Agency (EPA) and many states did not have adequate regulations governing the design, construction, monitoring and maintenance of sanitary landfills. It is because of this lack of regulation that, even after their closure, sanitary landfills will represent a widespread threat to surface and groundwaters for many years (Lee, Jones, and Ray, 1986). This threat comes from the contaminated liquid, or leachate, that is formed as water in the form of precipitation or groundwater flow percolates through the refuse and discharges to either subsurface or surface waters (Pohland, and Brunner, 1975). Once contamination has reached these bodies of waters there are many federal and state regulations, such as the Clean Water Act of 1972 and the Safe Drinking Water Act of 1974, which govern the concentrations of contaminants and set minimum national standards (Wentz, 1989).

What has been overlooked in the past is the importance and influence that stream and lake sediments have on the aqueous environment by acting as a temporary storage for contaminants. Many organic and metal contaminants are easily removed from the water column by adsorbtion onto suspended matter and settling to the bottom sediments. Therefore, while the surface water may meet the minimum water quality standards, the sediments may show an increase in concentration over time. These pollutants, however, are not permanently removed from the environment. Changes in the ambient conditions may cause certain pollutants to be released back into the overlying water long after they were initially affixed to the sediments (Shea 1988).

Although there are no EPA sediment quality standards or criteria at the present time, there have been studies undertaken to determine and model the ways and rates at which organic and metal contaminants, mainly from municipal and industrial point source discharges, are removed from the aqueous phase and accumulate on stream, lake and estuarine sediments. To determine the metal concentrations of sediments, various chemical extraction methods using nitric acid (Bloom and Ayling, 1977; Rule, 1986; and Mantei and Coonrod, 1989) or nitric acid and hydrochloric acid solutions (Capuzzo and Anderson, 1973; Trefry and Presley, 1976; and Schropp et al, 1990) have been used.

Cappuzo and Anderson (1973) investigated the distribution of chromium in the sediments of the Great Bay estuary in New Hampshire by taking cores above and below an industrial point source. Their study suggests that Cr settles out of the water column quickly and is sorbed to the sediments close to the discharge point. Gibbs (1973) described the major mechanisms of metal transport and attenuation in streams, such as metal adsorption onto suspended solids and precipitation on solids and/or in solution, and how these vary in importance with different metals. Filipek et al (1987) examined the pH effects on these mechanisms and found that certain metals such as Fe, Cu and Zn remain in solution in acidic waters.

Mantei and Coonrod (1989) studied the metal concentrations on the coarse grained sediment particles of streams affected by a sanitary landfill leachate. They investigated these larger grained particles, because of their lower transport capacity and higher residence time at certain locations, to determine if the metal concentrations would decrease with increased distance from the landfill. However, they found that these coarse grained particles were subject to transport by the streams and that there was no decreasing trend in concentrations.

Ramamoorthy and Rust (1989) studied the sediments of the Ottowa River for metal concentrations caused by pollution from pulp and paper mills. They found that the ability of sediments to sorb metals is increased by the presence of organic particles and by small mineral grains, and that desorption of metals from the sediments is increased when introduced to a high concentration NaCl solution.

LaBauve et al (1988) took 5 different soil samples and 2 different lake sediment samples and mixed them with a synthetically produced landfill leachate to determine the degree of metal retention on the soils and sediments. They determined that the lake sediments, due to their higher content of clays and iron oxides, retained more metals than the sandy soils. They showed that the pH of the soils and sediments was the most significant characteristic influencing metal retention.

A model by Davies-Colley et al (1984) shows how the uptake of the metals copper and cadmium varies in the different components of sediments. Adsorption to the iron hydroxides and organic matter was found to be greater than to the manganese hydroxides and clays.

Further studies by Ackerman et al (1983), Schneider and Weiler (1984), and Wong and Moy (1984) on sediment grain sizes have shown that the finer grained particles, clays and silts, have a greater affinity or binding capacity for metals than does sand. These studies also found that the presence of sands tends to dilute the total metal concentration of the sediment sample because of the normally low metal sorbing capabilities of the coarse sediment component.

The problems of separating the anthropogenic from the ambient influences of metals is discussed by Prohic and Juracic (1989). They suggest several methods to establish background levels, such as fossil sediments presumed to be a source rock, recent deposits in relatively unpolluted areas, and dated cores from the same location, using the concentration values from the deeper parts as background. Schropp et at (1990) determined the natural metal concentrations in clean estuarine sediments in Florida by normalizing the metals to a reference element, aluminum, to create a zone of natural concentrations. Metal concentrations from the contaminated Miami River and Biscayne Bay were then compared to the clean samples; metals which plotted outside the zones of natural concentrations were suggested to be anthropogenically enriched. Klinkhammer and Bender (1981), and Windom et al (1984) also normalized sediment metal concentrations to aluminum but then compared these values to average crustal abundances to show if the metal concentrations reflected the concentrations of the rocks from which they were weathered, or if another source was involved.

The purpose of the present study is to determine the impacts from a sanitary landfill to the sediments of a stream and pond system located in central New Hampshire. The Turnkey Landfill of Danbury, NH, which has been established as a source of organic and metal contaminants, is located in an abandoned sand and gravel pit just west of US route 4 and adjacent to Frazier Brook in west-central New Hampshire (Figure 1). This unlined landfill was in operation from 1976 to 1986 when it was closed and capped with a clay and vegetative cover. Possible environmental impacts were first indicated in the Spring of 1981 by the discoloration of Frazier Brook, approximately 100 meters west of the landfill. Hydrologic investigations were then begun to determine the extent and composition of the contamination at the landfill and the surrounding area, and to design a closure plan for the site (Gallup, 1982; Goldberg-Zoino Associates, Inc., 1989).

By 1984 the discoloration of Frazier Brook had extended to Eagle Pond approximately 2.4 km south, initiating an application of potassium permanganate and aluminum sulfate, ordered by the New Hampshire Water Supply and Pollution Control Commission, in order to precipitate the contamination out of the surface water (NHWSPCC,1984). Subsequent repetitive water samples from the brook and pond have shown a general decrease in the levels of organic and metal contaminants since the application in 1984 and the closure in 1986. The sediments of the brook and pond have, however, never been investigated.

Above the landfill area Frazier Brook is a relatively steep, shallow and fast moving pool and riffle stream. The stream bed consists of rocks and boulders with coarse grained sands and gravels. In the vicinity of the landfill where the brook meanders slowly through a large wetland, it is approximately 3 m wide by 1 m deep at bankfull stage. The streambed is mainly coarse sand with some silts and clays in areas where sediment deposition occurs, such as the insides of meanders (Dunne and Leopold, 1978).

The larger Walker Brook flows into Frazier Brook near the approximate midpoint of the landfill. From this confluence to Jack Wells Road, Frazier Brook is relatively straight, about 6-9 m wide by 0.5-1.5 m deep with a small flood plain and steep slopes that border the flood plain on the east. Below Jack Wells Rd to Eagle Pond, the brook is still deep and wide but has many more meanders. The bed is finer grained sand with mud in deep still pools.

Frazier Brook enters Eagle Pond from the north and exits to the south. The pond is approximately 200 m wide by 600 m long by 6 m deep. The inlet of Frazier Brook has created a flat, sandy delta which extends about 15 m into the pond where it drops off sharply. The shore surrounding the pond is mostly wooded with low sloping banks. A summer camp for children is located along the western shore. A section of the eastern bank is steep due to the use of rock and gravel fill to stabilize the bank for the adjacent railroad bed. The pond in this same area is shallow and rocky from this fill.

The bottom sediments of the first few meters from the edge of the pond are coarse grained sands and gravels to a water depth of 1-1.5 m. The bottom then slopes down to a maximum depth of approximately 6 m. The material of the sediments beyond the sand and gravel and at the base of the delta is fine grained, soft, black mud.

The drainage area of Frazier Brook at the inlet of the pond is approximately 22 square miles. The bedrock underlying the entire drainage area is the Cardigan pluton of Kinsman Quartz Monzonite, a quartz rich granitic rock with approximately equal portions of large potassium and plagioclase feldspar crystals (Lyons et al, 1986; and Gallup, 1982). The soils at the landfill site and along the length of Frazier Brook are mapped as the Colton-Rumney association and are described as being excessively drained sands and gravels in narrow valleys and in narrow, wet flood plains (United States Department of Agriculture, Soil Conservation Service, 1965).

To determine the extent of metal enrichment in the sediments, the groundwater flow and flux of metals through the landfill and into the brook was estimated. Grab samples of the bottom sediments in the brook and pond were collected in June and October, 1990. Three cores (A, B and C) from Eagle Pond were taken in June and August, 1990 and April, 1991 and sectioned intyo 1-2 cm intervals. All samples were dried and dissagregated, leached with Nitric Acid and analyzed by Inductively Coupled Plasma-Atomic Emission Spectrometry for the presence of 13 metals, Al, Fe, Mg, Mn, Si, As, Cd, Co, Cr, Cu, Pb and Zn. These metals were selected because, although landfill characteristics vary with the types and quantities of refuse, age of decomposition, and the nature of adjacent soils, many laboratory experiments have shown these metals to be the most commonly associated with landfill leachates (Pohland and Brunner, 1975; Chian, 1977; and van der Leeden, Troise and Todd, 1990).

To determine the background or ambient metal concentrations the values from the lower sections of Core B were normalized to aluminum and then compared to the other samples to assess the influence of the landfill. The third core was then analyzed by radionuclide counting procedures for ^{137}Cs and ^{210}Pb in order to estimate the sedimentation rate of the pond and to further define the impact of the landfill.

The grain size distribution and loss of ignition of the sediments were also determined to show the effects or correlation of metal concentrations to these sediment components.

II METHODS:

1) Groundwater and surface water regimes

To determine if the present metal loading of the sediments is solely a result of the operation of the landfill from 1976-1986 or is also a result of the continuing presence of the landfill since the closure in 1986 the subsurface and surface hydrology of the site was investigated. Utilizing data from monitoring wells and soil boring logs in the landfill, the long term average groundwater flow direction and rate were estimated. Long term average water table elevations from the eight monitoring wells surrounding the landfill (Figure 2) were used to determine the groundwater flow direction and hydraulic gradients through the two landfill sections (Heath, 1983). (See Appendix 1.1 for complete procedures).

The well and boring logs also provide the needed hydrologic parameters to estimate the groundwater flow rate through each section of the landfill via Darcys Law:

$$Q = kiA$$ (Freeze and Cherry, 1979)

where

Q = groundwater flow (m^3/d)
K = hydraulic conductivity (m/d)
i = hydraulic gradient (m/m)
A = cross sectional area (m^2) = $b*w$
b = avg. aquifer thickness (m)
w = aquifer width (m)

Table 1 and Figure 3 describe the data needed to determine these parameters. From Table 1, the long term average water table and bedrock elevations result in average aquifer thicknesses of 11.3 m (37.1 ft) and 13.5 m (44.3 ft) in the north and south areas respectively. The distances between monitoring wells 6L and 13, 283.5 m, (930 ft) in the north area and monitoring wells GZ-4 and GZ-2, 290 m, (950 ft) in the south area were used for the aquifer widths. A hydraulic conductivity value, determined from falling head permeameter tests (Gallup, 1982), of 15.2 m/d (50 ft/d) was used for both areas.

The groundwater seepage flux into Frazier Brook and the average hydraulic conductivity of the stream bed in the vicinity of the landfill was estimated using five seepage meters and piezometers (Lee and Cherry, 1978) see Figure 4. Seepage meters were constructed by drilling a 1/2 inch diameter hole in the bottom of an open ended 5 gallon plastic bucket, pushing the bucket into the brook sediments and then attaching a plastic bag to the hole. The volume of groundwater seeping through the stream bed and seepage meter was then collected and measured in the bag over time.

Piezometers were made by cutting slits in 3/4 inch diameter PVC pipes and wrapping nylon around the slits to prevent siltation. A 10 foot long, hollow cast iron pipe fitted with PVC tips was first hammered approximately 1 meter below the stream bed. The piezometers were then slipped into the pipe and held in place while the pipe was removed, leaving the piezometers a known depth below the stream bed. The vertical hydraulic gradient (i) was then calculated as the difference between the water level in the piezometer and the stream surface (^h) divided by the depth below the stream bed (L) (Lee and Cherry, 1978).

The groundwater flux into each seepage meter is calculated by:

$$v = V/t*A$$

where

v = seepage flux (cm/min)

V = volume inflow into bag (cm^3)

t = elapsed time (min)

A = area of seepage meter = 617.51 cm^2

The hydraulic conductivity of the stream bed is then calculated as:

$$K = v/(i)$$

where

K = hydraulic conductivity (m/d)

v = seepage flux (m/d)

i = vertical hydraulic gradient

The total groundwater seepage flux into this reach of Frazier Brook is therefore estimated by

$$Q = Kidw.$$

where

K = average hydraulic conductivity (m/d)

i = average vertical hydraulic gradient

d = stream distance between seepage meters = 823 m

w = average width of stream = 7.9 m

To augment the seepage meter procedure the brook was also gaged on April 20, 1991 with a digital stream velocity meter with the United States Geological Survey approved techniques (USGS, 1967). The brook was gaged at the farthest downstream seepage meter location (S5), at the inflow of Walker Brook into Frazier Brook, and at the upstream seepage meter location (S1) (see Fig 4). The difference in the stream flows at S5 and Walker Brook plus S1 approximates the flow of groundwater entering Frazier Brook throughout this reach. (See Appendix 1.3 for gaging data).

Frazier Brook was also gaged at the inlet and outlet of Eagle Pond in order to estimate the flow into Eagle Pond. From the outlet Frazier Brook flows about 3 kilometers into the Blackwater River. The USGS has maintained a stream gaging station on the Blackwater, from 1924 to the present, approximately 30 km from the Frazier Brook/Blackwater River confluence. The drainage area of the Frazier Brook inlet to Eagle Pond is 21.4 square miles (mi^2) and comprises 16.6% of the 129 mi^2 Blackwater River drainage basin. The inlet was gaged on April 26, 1991. five days after the last appreciable rain storm. and compared to the average daily flow of the Blackwater on the same date. Sixteen percent of the long term mean annual flow of the Blackwater River was then taken to approximate the mean annual flow of the Frazier Brook inlet. For comparison the flows of both streams were determined by the elevation to drainage area method (Dingman, 1978). This method utilizes the elevations at the point of interest and at the highest point in the drainage basin to estimate the long term mean annual flows of ungaged streams. The Dingman method (1978) was then used to estimate the mean annual flows of Frazier Brook and Walker Brook in the vicinity of the landfill.

2) Leachate plume boundaries and migration of plume over time.

The closure and capping of the landfill in 1986 was designed and constructed to prevent infiltration and percolation of rain and snow through the refuse and into the groundwater by channelling surface runoff into culverts and drainage swales and then into Frazier Brook (GZA, 1989). From personal observations, the cap has maintained its integrity but there is still visible leachate seeping into the wetland and into the brook along the length of the landfill.

Water quality data from the eight monitoring wells, from 1986 to November, 1990, were used to show the fluctuations of the plume downgradient of the landfill towards Frazier Brook over time. The water quality parameters available were chloride, iron and manganese. Chloride is an excellent tracer because of its conservative nature. The iron and manganese data were utilized since they were the only metals consistently analyzed over this time period (GZA, 1989).

Geophysical surveys using spontaneous potential and very low frequency (VLF) resistivity were used in December, 1990 and March and April, 1991 to show the migration and boundaries of the plume over the course of this study. Spontaneous potential utilizes two non-polarizing copper-copper sulfate electrodes and a high impedance millivolt meter to record the electrochemical potentials generated across boundaries of differing pore water composition (Corwin, 1988). The fixed-base configuration was used with the negative lead connected to the fixed base electrode, the positive lead to the advancing-measuring electrode. Measurements were taken at 10 meter intervals by placing the positive lead electrode connected to the millivolt meter into 10 cm holes dug into the ground surface. Survey lines ran along the length of the landfill, approximately perpendicular to the direction of groundwater flow, from 100 m north to 100 m south of the landfill. In the north area overhead power lines, culverts and visible metal were encountered which have been described as possible sources of interference or noise (Corwin, 1988). There were no such structures in the south area.

Very Low Frequency (VLF) is a method which utilizes low frequency radio waves generated from a global network of transmitting stations to determine the apparent resistivity of groundwater (Stewart and Bretnall, 1986). The primary field is produced by transmitters used for military communication which broadcast at frequencies between 18.8-24.0 kHz. At distances exceeding 500 miles (800 km) from the transmitter, the VLF field at the surface consists of a vertical electrical component, a horizontal electrical component directed radially away from the transmitter, and a primary horizontal magnetic field (Stewart and Bretnall, 1986). The Geonics EM-16R receiver measures the apparent resistivity and phase angle between the horizontal electrical and magnetic components which is measured across two probes inserted into the ground 10 meters apart. Flowing groundwater with dissolved contaminants will have a lower resistivity, and conversely a higher conductivity, than adjacent clean water. The phase angle gives information on relative depths of contamination; phase angles greater than 45 degrees indicates a low resistivity anomaly at greater depth. The VLF survey was run along the same SP survey line at 15.2 m (50 ft) intervals.

To sample and analyze the groundwater flowing from the landfill and entering Frazier Brook five additional piezometers were installed along the banks of the brook (see Fig 4 for locations). Samples were collected on April 20, 1991 and analyzed for the thirteen metals in accordance with EPA methodology (EPA, 1982). Three volumes of water from each piezometer were pumped before taking the sample. Approximately 0.1 ml of concentrated, reagent grade nitric acid was added to each sample bottle in order to lower the pH to below 2. The samples were placed in a cooler, brought back to the lab, filtered and refrigerated before being analyzed by ICP.

3) Contaminant loading in sediments

The first series of grab samples from the bottom sediments of Frazier Brook and Eagle Pond, and Core A from Eagle Pond were collected in June, 1990. Core B was taken in August and the second round of brook and pond grab samples were collected in October, 1990. A third core was taken on April 5, 1991. All grab samples were taken by hand by pushing a 7 cm diameter by 38 cm length of plastic core liner into the sediment approximately 5-10 cm, placing one core cap on the top extruded end to create a suction, removing the liner from the sediment and then placing a second cap on the bottom. Any water trapped in the liner was decanted and the sediment material then placed in labeled plastic bags.

The brook samples were collected from above the landfill to the inlet of Eagle Pond in locations of calm water, such as the insides of meanders or pools, where the velocity is lower and sediment deposition occurs (Dunne & Leopold, 1978). The pond grab samples were collected from the bottom by scuba diving to the bottom and using the core liner as described above.

Cores A and B were also taken by hand with a 7 cm diameter by 76 cm length of core liner in the same manner. Core C was taken by cutting a hole through the ice and lowering attached core liners down into the sediment. While pulling the liners back up through the ice, surface water trapped in the liners was siphoned off to prevent a pressure head from squeezing down on the core.

Each core was then brought to the shore, any remaining water decanted and the sediment core was extruded from the bottom of the liner onto a plastic covered board. The cores were then sectioned in the field with a nylon spatula into 1-2 cm intervals, and placed in labeled plastic bags. (See Figures 5 and 6 for sampling locations).

All samples were taken to the laboratory at the University of New Hampshire, the bags were opened in order to air dry the samples and covered with paper to prevent any fall out from entering the samples. To facilitate the drying process a large portion from each sample was placed in a clean petri dish and dried in a oven at 100 degrees centigrade (deg C) for 24 hours. Upon removal from the oven, the samples were dissagregated with a mortar and pestle and poured into clean, labeled 25 ml plastic vials for storage.

Metals bound or sorbed to the sediment particles were obtained by means of a Nitric Acid (HNO_3) leach. The concentrations of the metals were determined by Inductively Coupled Plasma Atomic Emission Spectrometry (ICP-AES). Approximately 2 grams from each sample was accurately weighed and placed in clean plastic 125 ml bottles. then leached with 25 ml of 10% volume/volume HNO_3 for 24 hours in order to dissolve the metals bound to the solid sediment particles. (See Appendix 2.1 for sample weights).

After 24 hours each sample was filtered into 25 ml plastic vials. A portion of each filtrate was then diluted by a factor of 10 by removing 2.0 ml of the filtrate by pipet and diluting with 18 ml of deionized water. The diluted samples were analyzed by ICP-AES for the presence of five major metals Al. Fe, Mg, Mn. and Si. while the undiluted samples were analyzed for eight trace metals As. Cd. Co, Cr, Cu, Pb, Ti. and Zn.

The ICP-AES analytical results were converted from parts per million (ppm) on a mass per volume (mg/l or ug/ml) basis into ppm on a mass per mass basis in the following way:

Trace Metals: (ppm ug/ml)*(25 ml)/sample wt = ug/g dry sediment

Major Metals: [(ppm)*(dilution factor of 10)*(25 ml)]/sample wt = ug/g dry sediment. (See Appendix 2.2).

The 24 hour HNO3 leaching procedure, carried out in batches of 20 samples, and the ICP-AES analytical procedure were tested by performing the same procedures on nitric acid and deionized water blanks and 2 randomly selected samples from each batch of 20. These replicate samples analytical results were then compared with the corresponding samples to estimate an error percent. (See Appendix 2.3).

To examine the effects of metals on different sediment components the grain size distribution was determined by wet sieving each sample from the first sediment sampling round. The loss on ignition was estimated by combusting approximately 3-5 grams of the brook and pond samples and 0.5-1.0 grams of Cores A and B sections in a muffle furnace at 550 C for approximately 2 hours. The material lost on ignition, or combustible matter, is related to the organic matter of the sediment (Capuzzo and Anderson. 1973). These two sediment components were then correlated to the metal concentrations individually and in a multiple regression. (See Appendices 2.4 and 2.5).

4) Eagle Pond sedimentation rate.

Core C was analyzed for ^{137}Cs and ^{210}Pb by non- destructive gamma spectrometry with a Canberra Germanium well detector coupled with a 4 K channel computer analyzer. After drying and dissagregating, approximately 2-3 grams of each section was lightly packed into 4 ml polyethylene omni vials and placed in the well detector for 8-20 hours, during which time the number of gamma decays of the two isotopes were counted

Cesium-137, first introduced into the atmosphere in 1953 by nuclear bomb explosions,, is quickly removed from the atmosphere by precipitation, and from surface water by adsorption to suspended materials which settle to the bottom sediments. The global fallout of ^{137}Cs reached a peak in 1963. If it is assumed that the activity of ^{137}Cs in the sediments is similar to the history in the atmosphere then the sedimentation rate of Eagle Pond can be estimated (Wong and Moy, 1984).

Analysis of Pb has also been used to estimate the sedimentation rate in lakes and estuaries (Farmer, 1977; and Windom et al, 1984). Lead-210, which comes from the radioactive decay of radium, is also quickly removed from the atmosphere and surface water and is bound to the sediments. The activity of ^{210}Pb, which decays at a half life of 23.1 years, at any depth below the sediment-water interface is therefore related to the time elapsed since deposition (Faure, 1986).

RESULTS AND DISCUSSION.

1) Groundwater and Surface water Regimes:

The groundwater flow direction, hydraulic gradient and flow rate through the landfill, which were determined using available data from monitoring wells installed in and around the landfill in 1982 and 1986 (GZA, 1989), show a variability due to seasonal water table fluctuations. Groundwater flow direction is least affected and is generally east to west through both sections of the landfill and perpendicular to Frazier Brook. As shown in Table 1 the water table elevations, generally high in Spring and low in Summer, affect the aquifer thickness (b) and hydraulic gradient (i) which are important components in determining the flow rate of an unconfined aquifer via Darcys Law (Freeze and Cherry, 1979). These parameters varied by 0.7 to 1.2 m and by 0.014 to 0.038 m/m which, therefore, resulted in variable flow rates of approximately 100 to 2000 cubic meters per day (m^3/d) through the north area and from 940 to 1500 m^3/d through the south area. Figure 7 shows the average groundwater flow direction and hydraulic gradients (Heath, 1978) used in conjunction with the data in Table 1 and Fig 3 to estimate average groundwater flow rates of 1200 and 1070 m^3/d through the north and south areas respectively.

The rates of groundwater flux into Frazier Brook as determined by the seepage meters and piezometers also show a seasonal variability (Table 2). The rate of groundwater entering this reach of the brook is calculated as:

$$Q = Kidw$$

where:

K = average hydraulic conductivity of the stream bed (m/d)
i = average vertical hydraulic gradient
d = length of stream reach (823 m)
w = average width of stream (7.9 m)

Table 2: Groundwater flux into Frazier Brook determined with seepage meters and piezometers.

	K (m/d)	i (m/m)	d (m)	w (m)	Q (m^3/d)	Q (m^3/s)
October 4, 1990	3.67	0.036	823	7.9	860	0.010
November 16, 1990	16.54	0.036	823	7.9	3870	0.045
April 20, 1991	11.27	0.046	823	7.9	3370	0.039
Average					2700	0.031

The values of (K) and (i) for the November data were averaged using 4 out of the 5 locations due to siltation in piezometer P5 which resulted in an erroneous vertical gradient measurement. This piezometer was subsequently purged for the April monitoring. The higher groundwater inflows in November and April were obtained during tree dormancy and suggest the importance of transpiration on the inflow into Frazier Brook. The average field measured influx of 2700 m^3/d or 0.031 m^3/s is slightly higher than the average groundwater flow rate through both sections of the landfill (2270 m^3/d or 0.026 m^3/d). This difference could of course be from field measuring error, as was the case with P5, or from the assumptions used in Darcys Law; ie uniform bedrock elevations. In an ideal hydrologic system, groundwater flows into a stream from both sides at equal rates (Heath, 1978). Figure 8 is a representation of this ideal system for Frazier Brook in which both sides or banks contribute an equal rate of the 2700 m^3/d inflow. These equal rates therefore suggest that approximately 60% of the flow under the landfill is entering the brook, while 40% is flowing under the brook to become part of the more regional groundwater flow regime.

The gaging of the brook on April 20 in the vicinity of the landfill was also used to estimate the groundwater flux into the brook by taking the difference between the downstream and upstream locations. As figure 4 shows the brook was gaged at the furthest downstream seepage meter (S5), at the inflow of Walker Brook, and at the upstream seepage meter (S1). The streamflow data, shown in Table 3, result in a difference of 0.045 m^3/s which is approximately 15% higher than the April 20 seepage meter estimate of 0.039 m^3/s. This difference could be from additional surface water input from the wetland located between the gaged points.

Table 3: Streamflow results and estimate of groundwater inflow of Frazier Brook along landfill reach.

Location		S5	Walker Brk.	S1	Difference
Streamflow	(m^3/s)	0.741	0.457	0.239	0.045
	(ft^3/s)	26.17	16.14	8.44	1.59

To determine the mean annual flow of Frazier Brook at the inlet of Eagle Pond the brook was gaged on April 26 and then compared to the average daily flow of the Blackwater River at the USGS gaging station on the same date (USGS, 1991). The flow at the inlet was determined to be 50 cubic feet per second (cfs) while the average daily flow of the Blackwater on the same day was 557 cfs, a difference of 9 %. It should be noted that the gaging of the inlet was a point measurement in time while the flow of the Blackwater was averaged over 24 hours.

As described, the drainage area of the Frazier Brook inlet to Eagle Pond is 21.4 square miles (mi^2) and comprises 16.6% of the 129 mi^2 Blackwater River drainage basin. From the USGS stream gaging station near Webster, NH the long term mean annual flow of the

Blackwater River is 214.9 cfs. If it is assumed that the Frazier Brook drainage area contributes 16.6% of this flow then the long term mean annual flow of Frazier Brook entering Eagle Pond is 35.7 cfs.

The mean flows of both streams were also determined by the elevation to drainage area method (Dingman, 1978) and estimated to be 193 cfs for the Blackwater and 31.4 cfs for Frazier Brook, a difference of 16.3%. The Dingman (1978) method was then used to estimate the mean annual flows of Frazier Brook in the vicinity of the landfill (Table 4).

The second column in Table 4 describes the mean annual flows based on the percent of drainage area to the Blackwater area. For example, the 21.4 mi^2 area of Frazier Brook is 16.6% of the 129 Blackwater basin and is therefore assumed to contribute 35.7 cfs towards the Blackwater flow. Comparing these values to the flows calculated by the Dingman method and to the flows determined by field gaging suggest that the long term mean annual flow of the Frazier Brook inlet to Eagle Pond is approximately 33 cfs and that Walker Brook contributes about one half of this flow.

Table 4: Stream Flow Data.

		Mean Annual Flows		Field Gaged	
	Drainage Area (mi^2)	% of Area (cfs)	Dingman (cfs)	4/20/91 (cfs)	4/26/91 (cfs)
Blackwater R.	129	214.9*	193	301*	557*
Frazier Brk inlet	21.4	35.7	31.4		50
Walker Brook	10.6	22.8	15.6	16.1	
Frazier Brk above Walker Brook	4.9	10.5	7.2	8.4	

*: From USGS stream gaging station near Webster, NH.

2) Leachate Plume Boundaries and Migration of Plume Over Time.

The water quality data for chloride, iron, and manganese from the monitoring wells around the landfill are plotted over time on Figure 9. In all three cases, although the general trend is a decrease in concentration in the downgradient wells since the closure in 1986, the concentrations are still above the values in the upgradient wells

(MW-9 & GZ-1). The variability of the concentrations over time is also shown, with low and high concentrations corresponding to the late fall/winter and spring/early summer months respectively. This suggests that the water table, rising from the spring snow melt, enters the bottom of the landfill and dissolves more contaminants and then continues to flow towards the downgradient wells and Frazier Brook.

The results of the geophysical surveys, (Figure 10 for plan view of survey lines) are shown in Figure 11. The initial SP field data were corrected for drift and smoothed using a 3-point running mean. The section 0-200 m was repeated on December 9, 1990 and March 21, 1991 and showed acceptable reproducibility with the same trends at identical points. Figure 11a is the SP profile along the landfill and suggests that in the north area the variability and generally positive SP values may be caused by noise from the power lines, culverts and metal which were encountered. The profile along the south area (approximately 600-900 m) shows a much more pronounced negative potential. Because there have been no published studies on SP applications involving delineation of contaminant flow it is difficult to compare the results from the Danbury Landfill. Instead the SP data can be interpreted in conjunction with the VLF results

Figure 11b is the profile generated from the VLF survey. It also shows variable resistivity in the north area (70-200 m), which suggest high concentrations of metallic wastes and other sources of interference (Stephens and Graham, 1985), followed by a decreasing trend along the remaining section of the north area. The area between the two landfill sections (450-550 m) and along the length of the south area is clearly shown by the peaks around 500 m and by the consistent lower resistivity values from 600-850 m, which have been shown to represent contaminated groundwater from a landfill source (Stephens and Graham, 1985; and Stewart and Bretnall, 1986). The increasing resistivity values beyond the 850 m point suggest the lower boundary of the plume via contaminant diffusion into the groundwater. Comparing the SP with the VLF profile suggests that both geophysical methods are measuring the leachate plume emitting from the south area and that the many sources of noise encountered in the north area are interfering with delineation of this plume.

Analysis of the phase angle profile from figure 11b further defines the plume characteristics. In the north area, angles are generally less than 45 degrees which suggests the lower resistivity anomaly is shallow or closer to the surface, where as in the south area the opposite is indicated. This appears to be consistent with the landfill cross sections profiled in Figure 3, which show a higher bedrock elevation in the north area. The cross sections also show the bedrock elevation rising along the lower part of the south area, which is consistent with the drop in the phase angle below 45 degrees beyond 800 m. This sloping bedrock coupled with the surface cap design may be influencing the groundwater to flow in a more northwesterly direction and preventing the plume from spreading more to the south.

The water quality results from the piezometers along the banks of the brook are shown in Table 5 and compared with the concentrations available from the upgradient monitoring wells and with EPA water quality standards. Although the data for the upgradient wells were taken from the last available field sampling round in November, 1990 and not all of the same metals were analyzed, comparison of the data further qualifies the leachate plume. Of the metals analyzed from the 2 upgradient wells and from the 5 piezometers all are found in greater concentrations in the piezometers downgradient of the landfill. Metal concentrations as determined by EPA water quality standards are also shown in Table 5. Of the metals analyzed, Fe and Mn are above the EPA standards, for the trace metals Cr, Cu, Pb and Zn are below.

Although there is variability in the metal concentrations found in the piezometers and only a few metals which were similarly analyzed from the upgradient wells the data suggests that the groundwater flowing into Frazier Brook contains high concentrations of metals generated from the landfill leachate.

3) Metal Loading in Sediments.

A. Frazier Brook Grab Samples.

Grab samples of the bottom sediments were taken in June, 1990 along the length of Frazier Brook, from above the landfill to the inlet and outlet of Eagle Pond. At each sampling location the attempt was made to collect the finer grained sediment due to the higher metal binding capacity of clays and silts than in sands (Wong and Moy, 1984 and Ackerman et al, 1983). The results of the ICP analyses from the first round of Frazier Brook sediment samples are shown in Figure 12 along with the percent error determined from the replicate samples.

The general trend of all the metals is to rise to a peak concentration at location #12, decrease in concentration to about #20 and then increase again to fairly constant concentrations in the downstream locations. Sample #12 was taken at the approximate midpoint of the landfill about 50 meters above the Walker Brook-Frazier Brook confluence in a small, shallow pond created by a beaver dam at the confluence. The sediment in this area is deep, fine grained mud. Below this point Walker Brook enters Frazier Brook increasing the flow by about one half. As described, from here to Jack Wells Road the brook is fast moving and fairly straight with few meanders and few areas where sediment deposition occurs. The sediment samples, therefore, consisted mainly of coarse sand with some silt and clay. The data suggest that the inflow from Walker Brook is preventing accumulation of fine grained, metal enriched sediment by keeping the material suspended through this reach.

Below Jack Wells Road to Eagle Pond (samples 6-1) the brook has many more large meanders and deep still pools. These are areas where sediment deposition occurs and where higher metal concentrations are found. Samples #1 and 21, taken 100 meters above and below Eagle Pond respectively, show the influence of the pond as a settling basin. All of the major metals except Mn show a decrease in concentrations in the outlet. Of the trace metals, Cd, Co, Cu, Pb and Zn are found in higher concentrations in the outlet. This may be due to additional affects of runoff from Eagle Pond Road and Route 4 (see Figure 1). These metals have been found to be associated with urban runoff and in high concentrations in stream water downstream of roads and bridges (Hoffman et al, 1985). This probable influence from road runoff is also seen in the Pb and Mn concentrations in samples 10 and 9 which were also taken in close proximity to Route 4.

To investigate the grain size effect on metal concentrations the percentages of fines (<63 um) and sands were determined by wet sieving each sample from the June round. The distribution of fines (Figure 13) follows the same trend as the metal concentrations; relatively high in the reach above Walker Brook, decreasing below Walker Brook and then increasing below Jack Wells Road. Since metals are found in higher concentrations in the fine grained particles and the presence of sands tends to dilute to total metal concentrations (Ackerman et al, 1983) the initial concentrations were normalized by the percent fines to determine if the higher initial analytical concentrations are representative or are associated with a high percent of fines. Normalizing the metal contents with percent fines in the sediment results in concentration values which can be better assessed and compared. If a high initial concentration is associated with a small percentage of fines, the resulting normalized value will be much larger and reflect enrichment in comparison to the situation where the concentration is associated with a high percent of fines as is the usual case.

The resulting [M]:Fines ratios vs. Sample #, plotted on Figure 14, show variable but relatively constant concentrations throughout the brook. The graphs indicate that the sediment sampled from Walker Brook to Jack Wells Road is as enriched in metals as the lower reach and that the low percentage of fines accounts for relatively high concentrations of metals.

Linear regression analyses were used to correlate the percentage of fines with metal concentrations and the percentage of combustible matter with metal concentrations. The significance of the relationships were tested using the statistical t-test to accept or reject the null hypotheses that there is no correlation between fines and metal concentration and no correlation between combustible matter and metal concentration. Rejecting the null hypothesis therefore indicates there is a correlation. Figure 15 is an example of the linear relationship between aluminum and the two independent variables. The strength of the relationship is reflected in the relatively high correlation coefficients (R) of 0.850 and 0.880 respectively. The results (Table 6) indicate there is a significant correlation between the amount of fines and metals except Mn and As, and between the amount of combustible matter and metals except As. The poor correlation with As is due to analyzed concentrations at zero or below detection limits. The outliers for Mn are samples 10, 9, 12, and 21 which may be showing enrichment from road runoff.

The two sediment components were then considered together in a multiple regression analysis. The null hypothesis was tested with the statistical F-test for multiple regression (Hamburg, 1985). The results of the metal concentrations vs. % fines plus % combustible matter (Table 7) indicate that combining the two components increases the strengh of the relationships for all metals and that, according to the F-test, there is a significant correlation between the two components and all metals except As and Cd. Although the individual linear correlations with Cd result in rejecting the null hypothesis, the low or weak correlation coefficients combine to suggest that Cd is not significantly associated with the amount of fines and combustible matter. The coefficient of multiple determination (R^2) values suggest that the amount of fines and combustible matter accounts for about 30 to 80% of the variability of the concentrations in the remaining metals.

The second round of sediment samples, from the same locations in the brook as well as additional points in the vicinity of the Walker Brook confluence, were collected in October, 1990 (see figure 5b) to further investigate the influence of Walker Brook and to determine if there were any seasonal affects or changes in the metal concentrations. Figure 16 shows the results of the October sampling round. The additional locations around sample #12 and above the confluence are points 24, 26, and 28. Sample 24 was taken 100 m upstream of June sample 11, directly across from the northern edge of the landfill, and further shows the upstream extent of metal enrichment. Number 28 was taken right above the beaver dam at the Walker Brook-Frazier Brook confluence. Again the general trend of higher concentrations in the vicinity of the north area above Walker Brook are followed by lower concentrations in the downstream locations and then again by an increase in the reach between Jack Wells Rd. and Eagle Pond. The concentrations in the three additional points are consistent with the June samples in the same section of the brook (samples 11 & 12) and further suggest the flushing capacity from Walker Brook.

To compare the June and October results the concentrations from both rounds are plotted on Figure 17. October sampling points 24, 26, and 28 have no corresponding values from June and have been removed. As can be seen the general trends and concentrations are consistent for each round. The major discrepancy is for sample 12 and the corresponding October sample 27 in which the concentrations of the major metals are much greater in June. In the trace metals this discrepancy is seen in Cr, Ti and Zn.

As can also be seen in Figure 17 the overall values from the October samples are generally less than the June samples. To determine if the values are significantly different, statistical tests using the Mann-Whitney test for two independent samples and the Wilcoxon test for paired samples were used (Dingman, 1991). The Mann-Whitney test combines the values from the two rounds and then sorts and ranks the data in ascending order. In the Wilcoxon test the values from the two rounds are paired or matched, the differences in the pairs are then calculated and sorted in ascending order. For both tests the null hypothesis of no difference or change in the metal concentrations in the June and October samples is tested by computing the standard normal variate (z) which is then compared to the critical z value for the chosen significance level. The results (Table 8) indicate that for the 90% confidence level (z-crit = 1.64) both tests suggest that the null hypothesis be accepted and conclude that there is no difference or change in the concentrations of Si, Cu, and Pb in the two sampling periods. The more stringent 95% confidence level (z-crit = 1.96) also shows no difference in these 3 metals, but slightly conflicting results for two of the major metals, Al and Mn. The results from the statistical tests in conjunction with the graphs suggest that there is a difference or a change in the concentrations of the major metals Al, Fe, Mg and Mn and of the trace metals Cd, Co, Cr, Ti and Zn. The change in concentrations from June to October of these metals may be due to changes in the ambient condition of the sediment and/or the water, such as pH or redox potential, as well as actual physical transport of metal loaded sediment by stream action.

3 B) Eagle Pond Grab Samples:

Metal concentration contours from the June, 1990 round of Eagle Pond grab samples were drawn assuming linear progression in concentration between sampling points, and are plotted on Figure 18 along with the percent error determined from replicate samples. The trend for the majority of the metals is to increase to higher concentrations in the middle of the pond. The exceptions are Mn and Co which have lower concentrations in the middle and increase towards the edges and the outlet of the pond. The major metals show more elongated and more widely distributed concentrations throughout the pond, with the higher concentrations extending beyond the middle of the pond towards the outlet. The trace metals seem to have the highest concentrations in a smaller area of the pond but also closer to the outlet.

The concentrations of metals in the pond sediments also show a progression or extension of the concentrations from the Frazier Brook samples. From the metal profiles of the brook sediment (Fig 12), the values along the lower reach (samples 1 & 2) are slightly lower but in the same range as the first contour plotted in Figure 18 for all metals. This suggests that the bulk of the metal loaded sediment is flowing through the brook and being deposited in the pond. This is consistent with the sediment distribution pattern in a stream and pond system in which the finer grains are held in suspension in the more turbulent stream before settling to the bottom of ponds or lakes (Dunne and Leopold, 1978).

Many of the trace metals also show relatively high concentrations in the north west area of the pond. This may be due to the high percentage of fines as shown in Figure 18f or from processes occurring at the summer camp. The normalized metal concentration to percent fines values (Figure 19) show the highest concentrations in this section of the pond as well as the middle and near the outlet and suggest enrichment in the sediment. The normalized contour plans also show an increase in concentrations from the inlet through the pond for all metals except Mg, Cr, Cu, and Pb. The data suggests that beyond the middle of the pond these metals are distributed more uniformly with respect to the amount of fines present.

The linear regression analyses of metal concentrations to percent fines and to percent combustible matter for the pond grab samples also show significant correlations. Figure 20 is an example of the linear relationship between aluminum and the two independent sediment components. Results of the statistical t-test (Table 9) indicate there is a significant correlation between the amount of fines and all metals except Mn, As and Ti, and between the amount of combustible matter and all metals except Mg, As, and Ti. Although the amounts of fine grains and combustible matter individually show no association with Mn and Mg respectively, the two components combine in a multiple regression to suggest a correlation (Table 10). When combined the two components account for 50 and 60% of the variability in the concentrations of Mg and Mn. The lack of individual correlations with As and Ti is also seen when the two components are combined.

The analytical results of the October samples were also contoured and are plotted on Figure 21. The concentrations of the metals are in the same range and show the same general trend as in the June samples but the distributions are not as widespread and appear more condensed in the middle of the pond. To determine if the values from the two time periods are significantly different the Mann-Whitney test was performed. As Figure 6 shows the samples from the two rounds were not taken in the same locations which prevents matching the June and October pairs for the Wilcoxon test. The results (Table 11) are similar to the Frazier Brook samples in that Si, Cu, and Pb show no significant changes in concentration. Aluminum and manganese are also shown to have undergone no changes in the pond sediments.

The data from the Mann-Whitney test as well as comparison of the contour maps (Figures 18 & 20) of the two sampling rounds suggest that there are significant differences or changes in the concentration and distribution of Fe, Mg, Cd, Cr, Ti, and Zn. The variability in the June and October contour maps for metals such as Al, Mn, Si, Co, Cu, and Pb may be attributed to errors in mapping the sampling locations exactly and therefore accurately measuring the distances between points and interpolating the contour lines, but the overall concentration values suggest consistent metal distributions in the two sampling rounds. Although the same types of errors are involved for the metals Fe, Mg, Cd, Cr, Ti, and Zn there appears to be an additional factor inherent in the discrepancies or variabilities in the distributions of these metals. The data suggest that this additional factor is the actual decrease in the concentrations of these metals in the bottom sediment from June to October. This decrease may have been caused by changes in the ambient conditions of the sediment or of the overlying water of Eagle Pond (ie: pH) which has been shown to be a significant factor in metal retention and in causing metals such as Fe and Zn to remain in solution (Filipek et al, 1987; and LaBauve et al, 1988).

3C) Eagle Pond Cores:

Cores A and B were taken in June and August, 1990 respectively, sectioned into 1-2 cm intervals and analyzed in the same manner for the same 13 metals. The analytical results were then plotted as profiles of metal concentration vs depth. Sediment particles loaded or sorbed with metals are continuously being supplied to the pond from Frazier Brook and also from direct runoff into the pond. Once in the pond these particles settle to the bottom and accumulate over time. The deeper sections therefore represent older material while the top sections of the core represent the most recently deposited material at the sediment/water interface. Analyzing and plotting metal concentrations vs. depth, therefore, creates a record of metal deposition over time.

The profiles from Core A (Figure 22) show a consistent pattern of high concentrations in the top most recent sections followed by decreasing concentrations to fairly constant levels in the deeper or older sediments. In the major metals Al. Mn and Si, the concentrations are fairly constant to a depth of 5-10 cm and then steadily decrease by 20-60% to about 25 cm. The profile of Fe also shows constant concentrations of approximately 35.000 ug/g to a depth of 5 cm followed by a sharp decrease of about 57% to 15,000 ug/g at 10 cm. a change of approximately 4000 ug/g per cm of sediment. Magnesium shows an increase in concentration with depth or conversely. in the older of deeper sections Mg is being removed or released from the sediment as time goes on.

The trace metals also show relatively high. constant concentrations. to core depths of 10-15 cm, followed by decreasing concentrations in the range of 50-70% to about 25-30 cm. Chromium and titanium display exceptions to this pattern, they are variable but fairly constant ranging from 10.5-12.5 and 420-450 ug/g respectively. The most dramatic change is shown in the Pb profile, decreasing from approximately 33 ug/g at a depth of 10 cm to about 1 ug/g at 30 cm, a decrease of 97% over 20 cm.

The variability displayed in the metal profiles may be caused by variability in the sedimentation rate and grain size distribution over time, which may have been influenced by the sand and gravel mining that has occurred along Frazier Brook. The grain size distribution of Core A, shown in Figure 23, indicates a fairly constant amount of fine grains of 85-90% to a depth of 10 cm, 80-85% to a depth of 25 cm and 75-80% to the bottom of the core with a sandy layer from 24-28 cm.

The normalized metal concentrations to percent fines profiles (Figure 24) show variable but constant values for Al, Mg and Si with little enrichment in these metals. The normalized profiles of Fe and Mn indicated enrichment in the sediment to a depth of 10-15 cm and then constant levels below. For the trace metals, Cd, Pb and Zn show enrichment with decreasing concentrations to a depth of 25-30 cm; Co and Cu indicate constant levels below 15 cm. Chromium and titanium are again fairly constant the entire length with Ti displaying a slight increase in concentration with depth. These normalized profiles suggest enrichment in the top 15-20 cm of sediment in the pond and constant or background levels below 25 cm.

Linear regression analyses between metal concentrations and the amount of fines and the amount of combustible matter again indicate significant correlations. Figure 25 shows the correlation between aluminum and the two independent components as an example. The linear regression results for Core A (Table 12) suggest a correlation between the amounts of fine grains and combustible matter and all metals except As and Ti. Again As is due to the measured concentrations at or below detection limits. The poor correlation with Ti in Core A is consistent with the Frazier Brook and Eagle Pond grab samples. The lack of correlation with As and Ti is maintained when the two components are combined in the multiple regression analyses (Table 13). For the remaining metals the two components combine to account for approximately 40 to 60% of the variability of the concentrations in the core.

The metal concentration vs. depth profiles for Core B (Figure 26) also show high concentrations, similar to Core A, in the top most recent sediments followed by decreasing concentrations with depth. For the major metals, Al and Si are variable but fairly constant around 15,000 and 2,600 ug/g respectively to a depth of 33 cm and then decreasing in the lower sections. The profile of Mg is again showing an increase in concentration with depth, or a release from the older sediments. Iron and manganese show the same trend of high concentrations in the top 5 cm followed by sharp decrease to about 13 cm, a slight decrease to about 25 cm and then fairly constant levels below this point. The trace metal profiles indicate the change in concentrations with depth much more consistently, with Cd, Cu, Pb and Zn showing the relatively high, constant levels extending to approximately 13 cm, and then the steady decrease in concentrations to a depth of 24 cm. Although the high concentration of Co extends to 9 cm and then decreases with depth there is a change in the slope at 13-14 cm and fairly constant levels below 24 cm. The profiles of Cr and Ti, as in Core A, are variable but relatively constant throughout the length of the core.

The concentrations of metals profiled in Core B are slightly greater than in Core A. This could be due to the relative locations in the pond, for, as shown on Figure 6, Core B was taken about 100 m further from the Frazier Brook inlet than Core A. This additional distance would allow a larger amount of the fine grain particles to remain in suspension before settling to the bottom. The higher percent of fines would, therefore, account for the relatively higher concentrations of metals in Core B.

The metal profiles from Core B also consistently show a change in slope to relatively constant concentrations below 24 cm. Since this section of Core B, from 24 to 42 cm, is deeper and sectioned into more intervals it has more of a consistent data base than Core A and is assumed, therefore, to represent the ambient or background concentrations of metals in the sediment. To determine the background levels for each metal, which can then be compared to the concentrations in the brook and pond samples, each metal was normalized to the conservative reference element Al. The concentrations of each metal (from 24-42 cm) were plotted against the corresponding values of aluminum to create a zone of ambient or background concentrations. The plots of metal concentrations vs Al along with the regression line and + - 95% confidence limits shown in Figure 27 suggest these background zones. Sediment which has not been anthropogenically enriched should therefore fall within the 95% confidence limits, while a sediment sample plotting outside these limits is suggested to be enriched in that particular metal (Schropp et al, 1990).

Results of regression analyses and t-test (Table 14) to test the null hypothesis of no correlation between metal and aluminum concentrations indicate there is a significant relationship between aluminum and all metals except As, Cd, and a poor correlation with Pb. Arsenic and cadmium have therefore been excluded from further analysis. The representative background concentration of Al ranged approximately 30% from 11,600-16,400 ug/g, while the concentrations of the other metals ranged between 25-45%. These metal to aluminum relationships are therefore used to compare the concentrations of metals in the brook and pond sediment in order to assess the extent of metal enrichment caused by the landfill leachate.

Metal concentrations vs aluminum for both rounds of the Frazier Brook sediment samples are plotted along with the regression lines and 95% confidence limits from the background data (Figure 28). Since the extent of aluminum concentration in the brook sediment, ranging about 85% from 1300-8300 ug/g, is much lower than the background zone the 95% confidence limits have been extrapolated to incorporate the lower Al range. As Figure 28 shows this lower aluminum range does correspond with lower concentrations of the other metals for the majority of samples. Those points which fall outside the extrapolated 95% confidence limits and suggest enrichment correspond to the highest points on the metal profiles shown in Figures 12 and 16. The actual samples or locations in the brook which are suggested to be enriched in metals are generally found in the samples above the Walker Brook confluence (June samples 11&12: October samples 24-28) and in the reach from Jack Wells Road to Eagle Pond (June samples 6-1: October samples 39-44).

For the major metals, Fe, Mg, and Mn, these enriched samples, approximately 50-80% of the June samples decreased to 30-60% of the October samples. All samples from both rounds indicate enrichment in Si.

The trace metals to aluminum correlations suggest that the brook sediment is not enriched in Cr and Cu but highly enriched in Co and Ti. Cobalt had 18 of the 21, or 86% of the June samples decrease to 15 of the 24 or 63% of the October samples, while Ti went from 100% to 92% of the samples showing enrichment. The few samples which show enrichment in lead are those which were taken in close proximity to the roads in the area and may be showing the influence of road runoff. The zinc correlation data indicates that the concentrations in the few enriched June samples decreased to ambient concentrations in October. This is consistent with the results from the Mann-Whitney and Wilcoxon statistical tests which suggested that zinc decreased in concentration over the two sampling rounds.

The metals vs aluminum concentration correlations for both rounds of the Eagle Pond bottom samples are plotted in Figure 29. The Al concentrations in these samples range approximately 89% from 2800-17500 ug/g and encompass the background zone while the concentrations of the other metals generally range from 70-90%. Lead has the greatest range or fluctuation in concentration, approximately 98% from 1.0 to 43.9 ug/g.

Assessing the metal enrichment of the pond sediments by aluminum normalization indicates that the sediment is enriched in different metals in different areas of the pond. The main exception is chromium which like the Frazier Brook sediment falls within the extrapolated background zone in all samples throughout the pond and is therefore suggested to be in the range of ambient concentrations.

For the major metals, Fe and Mn are enriched in all samples throughout the pond except in those from around the inlet, with the relatively highest enrichment seen in the middle of the pond. Magnesium and silica are generally enriched in those samples which have a low percent of fine grains, or a high amount of sand. Silica is enriched throughout the pond, except in the middle and especially near the inlet around the sandy delta. Magnesium is also enriched near the inlet and in the area along the western shore.

The trace metal plots suggest that Co is enriched in the sediment throughout the pond, while Cr is within background or ambient concentrations. Copper, lead and zinc are shown to be enriched in all areas of the pond except the inlet, while Ti is enriched throughout the pond including the inlet area.

The data further support the relationship between metal concentration and the amount of fine grained clays and silts. Magnesium, silica and titanium are found to be enriched in the samples near the inlet where the inflow from the brook has created the large sandy delta. The remaining metals, although enriched throughout the pond, are shown to have the relatively highest enrichment in the middle of the pond, which corresponds to the highest percent of fine grained sediment.

The data also indicate that, although the sediment is enriched throughout the pond, the relatively highest distribution is located mainly in the middle and western shore areas. This could be due to the configuration of the inlet and the shore line (see Figures 18 and 21) which may be creating a bottleneck effect and causing the accumulation of metal loaded sediment to be contained in this area. The enrichment seen in the western shore area may also be from additional inputs of metals from the summer camp located along this shore.

These metals to aluminum correlation analyses reveal certain consistent trends in the brook and pond sediments. For example chromium is shown to be within background or ambient concentrations throughout both the brook and pond sediments. The data also point out that for each metal there is a higher percent of the June samples which fall outside the extrapolated background zone, which suggests enrichment, than the October samples in both the brook and pond sediment. There is also a larger percent of samples suggested to be enriched in the pond than in the brook. For example, copper went from 0% of the samples in the brook to approximately 80% of the pond samples showing enrichment. This increase in the pond sediments, which further defines the ponds influence as a settling basin for metal enriched sediment, is seen in all metals except Mg, Si and Ti. The extent of enrichment of these three metals actually declined in the pond, for example Si went from 100% of the brook samples to 80% of the pond samples showing enrichment. Again the data indicate that these three metals are enriched in those sediment samples which have a high percent of sand.

The metal vs aluminum plots from Cores A and B (Figure 30) show the general trend of the top sections of both cores being enriched in metals while the lower sections fall within background levels. The majority of points from Core B which fall within each background metal zone are of course those sections which were used to generate the zones (24-42 cm). The points or sections from each core which fall outside the +- 95% confidence limits describe the depths in the sediment in which enrichment first occurred.

The Fe to Al correlation plot indicates that in both cores the sediment is enriched from 0 to 20 cm, for Mn the sections from Core A which indicate enrichment are from 0-19.5 cm and from 0-21 cm in Core B. The analysis of Mg suggests that because of the increasing concentration with depth, as shown in Figure 22, the first 8 cm of Core A are within background levels and that the lower sections, from 9-37 cm, are enriched in Mg. The increasing concentration with depth in Core B (Figure 26) results in the sections 14 to 42 cm being within background concentrations while the top of the core, sections 0-13 cm, is below the background levels or undersaturated in Mg. The Si vs Al correlation plot indicates that for Core A the top 5 cm and from 20-34 cm are enriched while the sections from 6-19 cm are within the background levels. The lower sections of Core A which are shown to be enriched in Si correspond to the sandy layer shown in the grain size distribution vs depth profile in Figure 23. this is consistent with the analyses from the brook and pond sediments which also suggested enrichment in those samples with a relatively high amount of sand. The analysis of Si from Core B suggests that the sections 0-6 cm and 19-42 cm are within background levels while the sections 9-18 cm are below ambient concentration or undersaturated.

The trace metals to aluminum plots suggest that Cr is again within background levels in both cores and that the sections which fall outside the background limits and suggest enrichment occur at different depths in the two cores. Cobalt and zinc are both shown to enriched in the sections 0-28 cm in Core A and 0-23 cm in Core B. Copper is enriched from 0-19.5 cm in Core A and from 0-22 cm in

Core B, while Pb is enriched from 0-18 cm in Core A and from 0-23 cm in Core B. The analysis for Ti suggests that all of Core A is enriched while all of core B is within background concentrations.

The aluminum normalization analyses show that Core A has relatively higher concentrations of Mg, Si and Ti than Core B. Those sections which are shown to be enriched in these metals in Core A correspond to the sections in Core B which are within background concentrations while those sections from Core A that fall within background levels correspond to sections in Core B that are below background levels. This is consistent with the Eagle Pond grab samples analyses which showed that the area of the pond with the higher levels of enrichment in these three metals is in the northern half of the pond which is the area that Core A was taken.

4) Eagle Pond Sedimentation Rate:

The third core from Eagle Pond, taken on April 4, 1991 was sectioned into 1-2 cm intervals to a depth of 40 cm and then into 3 cm sections to a total recovered depth of 52 cm. Approximately 2 grams from each dried, dissagregated section were analyzed by radionuclide counting procedures for the presence of ^{137}Cs and ^{210}Pb in order to estimate the rate of sediment accumulation in the pond and to correlate time with depth in all three cores. Both isotopes are quickly removed from the atmosphere by precipitation and from the water column by adsorbing onto fine grain particles and settling to the bottom sediments where they continue to undergo radioactive decay with a half life of 30.1 yr for ^{137}Cs and 22.3 yr for ^{210}Pb (Wong and Moy, 1984; Robbins and Edington, 1975).

The sedimentation rate of lakes and estuaries can be estimated using ^{137}Cs by assuming that the accumulation pattern in the sediments reflects the well known history of global atmospheric fallout which first became detectable in 1954 with the introduction of nuclear bomb tests, and reached a maximum in 1963. The results of the ^{137}Cs analysis (Figure 31) indicate that the isotope is first detected at 18 cm at an activity of 0.827 disintegrations per minute per gram of sediment (dpm/g). The activity then increases steadily upward in the core to a peak concentration of 2.492 dpm/g at 13 cm, followed by a sharp decrease and then variable but fairly constant concentrations to the sediment/water interface. If this profile is interpreted in the classical way then the depth at which ^{137}Cs is first detected (18 cm) can be considered to be 1954 and the top of the core can be considered the time of sampling or 1991. The sedimentation rate is therefore estimated to be 18 cm/37 yr = 0.49 cm/yr. When used in conjunction with depth, this rate suggests that the peak concentration at 13 cm occurs in 1964 which, although slightly different than the recorded history of ^{137}Cs, is an excellent first estimate.

To determine the sedimentation rate using ^{210}Pb the total activities vs depth are first plotted on Figure 32. The profile shows a zone of variable but fairly constant activity to a depth of approximately 10 cm, followed by a sharp decrease in activity with depth, to about 19-20 cm, that suggests exponential decay. Below 19 cm the activity is again variable but fairly constant around 3.0 dpm/g and is therefore assumed to represent the amount of ^{210}Pb that is supported by the radioactive decay of radium in the sediments. This average activity of 3.263 dpm/g was then subtracted from the total activities of the top sections of the core to derive Excess ^{210}Pb from which the sedimentation rate was determined (Windom et al, 1983; Farmer, 1978).

Figure 33 is the semi-log plot of Excess ^{210}Pb vs depth. From the slope of the best fit exponential line the sedimentation rate (a) is calculated by:

$$a = -l/m \text{ (Faure, 1986)}$$

where

l = decay constant of ^{210}Pb = $-0.0311\ yr^{-1}$

m = slope = -0.041

The sedimentation rate is therefore calculated to be 0.76 cm/yr which is about 50% higher than the rate estimated from the ^{137}Cs data.

It has been shown that sedimentation rates determined in this manner may be artificially high due to the large percentage of water, by volume, in the top sections labelled with Excess ^{210}Pb. These top sections are not as compacted as the lower sections but become increasingly so with depth in the sediment column until a constant water content is obtained (Farmer, 1978; Robbins and Edington, 1975). To take the compaction of the Eagle Pond sediments into account the percent water content on a mass basis was determined from the initial wet and dry weight data from each section. The percent water content by volume or porosity of the core was then determined using densities of 1.0 and 2.65 g/cm^3 for the water and the sediment respectively. The results, Table 15 and Figure 34, show relatively high values for water content and porosity in the top uncompacted sections decreasing with depth to fairly constant values of approximately 73% and 88% respectively. The sharply decreasing porosity around 20 cm suggests a sandy layer or lense which is consistent with the grain size distribution of Core A profiled in Figure 23. The Wet:Dry volume ratios were then derived and plotted in order to normalize the uncompacted sections to the compacted sections. As shown in Figure 34 the Wet:Dry ratios approach a fairly constant value of 8.17 in the lower compacted sections. This average value was then divided by the ratios of the uncompacted sections. This procedure converts the measured length of an uncompacted section (1 cm) to the assumed length at full compaction.

The compaction corrected depths were then plotted vs Excess ^{210}Pb in Figure 35. The sedimentation rate, derived from the slope (0.049), is then calculated to be 0.63 cm/yr, which is still about 20% higher than the rate determined with the ^{137}Cs data. This discrepancy between the two rates may be a result of the top few centimeters of the sediment being disturbed and not collected in the core. But when the standard error of the slope (0.013) is taken into account the ^{210}Pb compaction corrected rate ranges from 0.50 to 0.86 cm/yr which encompasses the ^{137}Cs rate.

These sedimentation rates of 0.50 and 0.63 cm/yr were then used in conjunction with the concentrations vs depth profiles of Cores A and B. The rates suggest that the time frame of the landfill, 1976-1986, corresponds to a range of depth of 9.5-2.5 cm. This time-depth range was then transposed onto the profiles of Cores A & B to further assess the impacts of the landfill on the sediments of Eagle Pond. Figure 36 shows a general trend of slightly increasing concentrations in the cores before the opening of the landfill followed by a sharp increase around 9-10 cm depth.

For the major metals this increase in concentration around the opening of the landfill in 1976 is seen most dramatically in the Fe profiles which have increases of approximately 130% (15,000 to

35,000 ug/g), and 76% (17,000 to 30,000 ug/g), in Cores A and B respectively. Manganese and silica also show increasing trends over the life of the landfill, increasing by about 40 and 20% respectively. The aluminum profile shows the variable but fairly constant concentrations before the landfill continuing during the landfill. The profile of Mg shows a continuous decrease in concentration with a slight increase during the early years of the landfill. All of the major metals show a trend of decreasing concentrations since the closure of the landfill in 1986 but still above pre-landfill levels.

For the trace metals only Co and Zn show increasing concentrations corresponding to the opening of the landfill, with the changes occurring in the early years followed by fairly constant levels in the later years of the landfill. Cadmium and chromium appear to be unaffected by the presence of the landfill; Cd maintaining the increasing concentration trend while Cr remaining constant before, during and after the landfill. The concentrations of Cu and Pb appear to have actually decreased in the sediments since the opening of the landfill. The profiles of Cu and Pb show high rates of increasing concentrations in the pre-landfill sediments which then remain fairly constant and slightly decreasing in Core A over the life of the landfill. Titanium is showing an overall decreasing trend with a small increase corresponding to the early years of the landfill.

The analysis suggests that even though there were trends of increasing concentrations in the sediments before the landfill began operating, these concentrations began increasing at higher rates with the opening of the landfill. The pre-landfill sediments may be reflecting the input of metals into the brook and then the pond from the sand and gravel mining at the landfill and other sites along Frazier Brook Once the landfill was opened, the input of metals was greatly enhanced and is reflected in the change in slopes of the concentration profiles. The affects of the closing and capping of the landfill in 1984-1986 are also seen by the generally decreasing trends in the profiles. Because the sedimentation rate is high and Cores A & B were taken so relatively soon after the closure it is difficult to assess the future trends in the metals concentrations. It does appear however that closing and capping the landfill has done a great deal in slowing the influx of metals into the brook and therefore into the pond. The field investigation data from the seepage meters and piezometers suggest that there is leachate still being generated from the landfill, dissolving into the groundwater and entering Frazier Brook. Once in the brook these metals are available to be sorbed to fine grained particles, transported by the brook and deposited in the pond sediments. The data also suggest that although the metal concentrations in the upper sediments are showing decreasing trends since the closure of the landfill, the concentrations are still at levels which may pose a significant threat to the water quality should the ambient conditions change and cause the metals to desorb from the sediments and be released back into the water column.

CONCLUSIONS:

Investigations of water quality trends from eight remaining monitoring wells around the landfill suggest that landfill leachate is still being generated and dissolved into the groundwater with concentrations of metals that, although decreasing since the closure in 1986, are above the upgradient or background levels. Field data from seepage meters and piezometers suggest that approximately 60% of the groundwater flowing under the landfill is being discharged into Frazier Brook at an average rate of 0.031 m/s while 40% is flowing under the brook to become part of the more regional groundwater flow regime. The field data also indicates that the groundwater discharging into the brook contains metals at concentrations that are above background levels and EPA standards.

Once discharged into the brook the metals are available to be adsorbed to the fine grained and organic components of sediments and transported downstream. The metals to aluminum correlation analyses for the two rounds of brook samples indicate that the sediment is generally enriched in metals in the reaches above the Walker Brook-Frazier Brook confluence and from Jack Wells Road to Eagle Pond. These are the reaches of Frazier Brook that have more meanders and more locations for sediment deposition. The data further suggest that all of the samples are enriched in Si and Ti, while none are enriched in Cr or Cu. Statistical analyses suggest there was a decrease in metal concentrations in the sediments from June to October in all metals except Si, Cu, and Pb.

The concentrations of metals in the Eagle Pond bottom sediments are shown to be initially distributed throughout the pond with the highest concentrations generally found in the middle of the pond. When normalized or referenced to the aluminum concentrations the enrichment of the sediments is shown to occur in the middle and western portions of the pond for all metals except Cr which is shown to be within background levels. The statistical analyses regarding concentration changes in the pond sediments suggest there was a decrease in the concentrations of Fe, Mg, Cd, Cr, Ti and Zn. Of the remaining metals which were shown to have undergone no significant changes in concentrations over the two sampling periods, Si, Cu, and Pb are consistent with the results from the brook samples. Comparing the concentrations, distributions and enrichment of metals in the brook and pond sediments together reveal that the pond is acting as a settling basin for metal loaded sediment flowing from the brook.

The metals to aluminum concentrations analyses of Cores A and B suggest that the depths of metal enrichment begin approximately 18-24 cm below the sediment water interface for all metals except Cr which, like the brook and pond samples, is shown to be within ambient levels. Utilizing the sedimentation rates estimated from the ^{137}Cs and ^{210}Pb analyses, 0.50 and 0.63 cm/yr respectively, the 10 year time frame of the landfill was superimposed onto the metals vs depth profiles of Cores A and B. The results indicate there were increasing concentrations of metals being input into the pond prior to the opening of the landfill but that the rates increased once the landfill opened. The data suggest that the landfill has increased the sediment concentrations of major metals Fe, Mn and Si with little or no influence on the overall trends of Al and Mg. For the trace metals analyzed the landfill appears to have increased the concentrations of Co and Zn with no apparent influence on Cd, Cr, Cu, Pb and Ti.

References Cited:

Ackerman, F., H. Bergmann, and U. Schleichert, 1983. Monitoring of heavy metals in coastal and estuarine sediments-aquestion of grain-size: <20 um versus <60 um. Environmental Technology Letters, v. 4, pp. 317-328.

Bloom, H. and G.M. Ayling, 1977. Heavy metals in the Derwent Estuary. Environmental Geology, v. 2, no. 1, pp. 3-22.

Capuzzo, J.M., and F.E. Anderson, 1973. The use of modern chromium accumulations to determine estuarine sedimentation rates. Marine Geology, v. 14, pp. 225-235.

Chian, E.S.K., 1977. Stability of organic matter in landfill leachates. Water Research, v. 11, pp. 225-232.

Corwin, R.F., 1988. The self-potential method for environmental and engineering applications. Society for Exploration Geophysicists, unpublished.

Davies-Colley, R.J., P.O. Nelson, and K.J. Williamson, 1984. Copper and cadmium uptake by estuarine sedimentary phases. Environmental Science and Technology, v. 18, pp. 491-499.

Dingman, S.L., 1978. Synthesis of flow-duration curves for unregulated streams in New Hampshire. Water Resources Bulletin, v. 14, pp. 1481-1502.

Dingman, S.L., 1991. Statistical Inference, personal communication.

Dunne, T., and L.B. Leopold 1978. Water in environmental planning. San Francisco: W.H. Freeman and Company. 818 pp.

Farmer, J.G., 1977. The determination of sedimentation rates in Lake Ontario using the Pb method. Canadian Journal of Earth Sciences. v. 15, pp. 431-437.

Faure, G., 1986. Principles of Isotope Geology, second edition. New York: John Wiley & Sons. 589 pp.

Filipek, L.H., D.K. Nordstrom , and W.H. Ficklin, 1987. Interaction of acid mine drainage with waters and sediments of West Squaw Creek in the West Shasta Mining District, California. Environmental Science and Technology, v. 21, p. 388-396.

Freeze, R.A., and J.A. Cherry 1979. Groundwater. New Jersey: Prentice-Hall, Inc. 604 pp.

Gallup, M.L., 1982. Hydrogeologic evaluation, Turnkey landfill of Danbury, South Danbury, New Hampshire

Gibbs, R.J., 1973. Mechanisms of trace metal transport in rivers. Science, v. 180, p. 71-73.

Goldberg-Zoino Associates, 1989. Evaluation of post closure water quality data, Turnkey Landfill of Danbury, Danbury, N.H.

Hamburg, M., 1985. Basic statistics a modern approach, third edition. San Diego: Harcourt Brace Jovanovich, Publishers. 548 pp.

Heath, R.C., 1983. Basic ground water hydrology. USGS water supply paper 2220. Prepared in cooperation with North Carolina Department of Natural Resources and Community Development. p. 11.

Hoffman, E. J., J.S. Latimer, C.D. Hunt, G.L. Miller, and J.G. Quinn, 1985. Stormwater runoff from highways. Water, Air, and Soil Pollution, v. 25 no. 4, pp. 349-364.

Klinkhammer, G.P. and M.L. Bender, 1981. Trace metal distributions in the Hudson River estuary. Estuarine, Coastal and Shelf Science, v. 12, pp. 629-643.

LaBauve, J.M., J. Kotuby-Amacher, and R.P. Gambrell, 1988. The effect of soil properties and synthetic municipal landfill leachate on the retention of Cd, Ni, Pb, and Zn in soil and sediment materials. Journal Water Pollution Control Federation, v. 60, pp. 379-385.

Lee, D.R. and J.A. Cherry, 1978. A field exercise on groundwater flow using seepage meters and mini-piezometers. Journal of Geological Education, v. 27, pp. 6-10.

Lee, G.F., R.A Jones and C. Ray 1986. Sanitary landfill leachate recycle. Biocycle, v. 27, no. 1, pp. 36-38.

Lyons, J.B., W.A. Bothner, R.H. Moench, and J.B. Thompson Jr. (eds.), 1986. Interim geologic map of New Hampshire: N.H. Dept. of Environmental Services Map OF-86-1, scale 1:250,000.

Mantei, E.J., and D.B. Coonrod 1989. Heavy metal content in the stream sediments adjacent to a sanitary landfill. Environmental Geology and Water Science, v. 13, p. 51-58.

New Hampshire Water Supply and Pollution Control Commission, 1984. Eagle Pond watershed water quality sampling results, 1983 and 1984.

Pohland, G.F., and D. Brunner 1975. Sanitary landfill stabilization with leachate recycle and residual treatment. Final Report to Solid and Hazardous Waste Research Div., Municipal Environ. Res. Lab., U.S. EPA, Cincinnati, Ohio.

Prohic, E., and M. Juracic, 1989. Heavy metals in sediments-Problems concerning determination of the anthropogenic influence. Study of the Krka River Estuary, eastern Adriatic coast, Yugoslavia. Environmental Geology and Water Science, v. 13, p. 145-151.

Ramamoorthy S., and B.R. Rust 1978. Heavy metal exchange processes in sediment-water systems. Environmental Geology, v. 2, pp. 165-172.

Rule, J., 1986. Assessment of trace element geochemistry of Hampton Roads Harbor and lower Chesapeake Bay area sediments. Environmental Geology, v. 8, no. 4, pp. 209-219.

Schneider, B. and K. Weiler, 1984. A quick grain size correction procedure for trace metal contents of sediments. Environmental Technology Letters, v. 5, pp. 245-256.

Schropp, S.J., F.G. Lewis, H.L. Windom, J.D. Ryan, F.D. Calder, and L.C. Burney, 1990. Interpretation of metal concentrations in estuarine sediments of Florida using aluminum as a reference element. Estuaries, v. 13, no. 3, pp. 227-235.

Shea, D. 1988. Developing national sediment quality criteria. Environmental Science and Technology, v. 22, no. 11, pp. 1256-1261.

Stephens, L.E., and B.W. Graham, 1985. An electromagnetic survey over the Gloucester Landfill site for the detection of contaminated groundwater. Geological Survey of Canada, pp. 431-440.

Stewart, M., and R. Bretnall, 1986. Interpretation of VLF resistivity data for groundwater contamination surveys. Groundwater Monitoring Review, v. 6, no. 1, pp. 71-75.

Trefry, J.H., and B.J. Presley, 1976. Heavy metals in sediments from San Antonio Bay and the northwest Gulf of Mexico. Environmental Geology, v. 1, no. 5, pp. 283-294.

U. S. EPA, 1982. Test method. Inductively coupled plasma-atomic emission spectrometric method for trace element analysis of water and wastes-Method 200.7. Environmental Monitoring and Support Laboratory, Cincinnati, OH.

U.S. EPA, 1984. National Priorities List. First Edition.

U.S. EPA Office of Water-Waste Management, Office of Solid Waste, and F.C. Hart 1980. Technology, prevalence and economics of landfill disposal of solid waste. Solid Waste Management Series, U.S. Office of Solid Waste.

United States Department of Agriculture, Soil Conservation Service, 1965. Soil Survey Merrimack County, New Hampshire. Washington, DC: US Government Printing Office. 94 pp.

United States Geological Survey, 1991. Daily streamflow data for Blackwater River near Webster, NH, station number 01087000. Water year October, 1990-September, 1991. Unpublished.

United States Geological Survey, 1967. Techniques of water-resources investigations of the United States Geological Survey. Department of the Interior, Geological Survey: Washington, D.C.

van der Leeden, F., F.L. Troise and D.K. Todd, 1990. The water encyclopedia, second edition. Chelsea, Michigan: Lewis Publishers.

Wentz, C.A., 1989. Hazardous waste management. New York: McGraw-Hill Publishing Company. 461 pp.

Windom, H.L., S. Silpipat, A. Chanpongsang, R.G. Smith, Jr., and M. Hungspreugs,, 1984. Trace metal composition of and accumulation rates of sediments in the upper Gulf of Thailand. Estuarine, Coastal and Shelf Science, v. 19, pp. 133-142.

Wong, G.T.F., and C.S. Moy, 1984. Cesium-137, metals and organic carbon in the sediments of the James River estuary, Virginia. Estuarine, Coastal and Shelf Science, v. 18, pp. 37-49.

LIST OF TABLES:

TABLE 1

Groundwater Flow Through Landfill (data from Gallup, 1982; and GZA, 1989).

NORTH AREA:	Water Table Elevations: (meters m.s.l.)						Long Term
	Aug. 86	Nov. 86	Jan. 87	Apr. 87	Jul. 90	Nov. 90	Average
6L	203.3	203.1	203.3	203.7	203.2	203.4	203.3
7L	203.2	203.0	203.1	203.5	203.1	203.3	203.2
13	203.2	202.7	203.2	203.5	203.0	203.7	203.2
Avg.	203.2	202.9	203.2	203.6	203.1	203.5	203.3
Bed. Elev.	192.0	192.0	192.0	192.0	192.0	192.0	192.0
b (m)	11.2	10.9	11.2	11.6	11.1	11.5	11.3
i (m/m)	0.002	0.029	0.031	0.040	0.029	0.019	0.025
K (m/d)	15.2	15.2	15.2	15.2	15.2	15.2	15.2
w (m)	283.5	283.5	283.5	283.5	283.5	283.5	283.5
Q (cmd)	97	1,362	1,496	1,999	1,387	942	1,217

SOUTH AREA:		Water Table Elevations: (meters m.s.l.)						Long Term
	Apr. 86	Aug. 86	Nov. 86	Jan. 87	Apr. 87	Jul. 90	Nov. 90	Average
GZ-4	203.5	202.4	202.2	202.4	203.0	202.2	202.8	202.6
GZ-3	202.1	202.3	202.0	202.1	202.7	201.1	202.3	202.1
GZ-2	202.6	201.4	201.3	201.5	202.8	201.3	201.8	201.8
Avg.	202.7	202.0	201.8	202.0	202.8	201.5	202.3	202.2
Bed. Elev.	188.7	188.7	188.7	188.7	188.7	188.7	188.7	188.7
b (m)	14.0	13.3	13.1	13.3	14.1	12.9	13.6	13.5
i (m/m)	0.017	0.016	0.020	0.019	0.017	0.027	0.013	0.018
K (m/d)	15.2	15.2	15.2	15.2	15.2	15.2	15.2	15.2
w (m)	290	290	290	290	290	290	290	290
Q (cmd)	1,049	938	1,155	1,114	1,057	1,535	779	1,071

Q = groundwater flow = Kibw
Bed. Elev. = Bedrock elevation in meters above mean sea level
b = aquifer thickness = (water table elev. - bedrock elev.)
i = hydraulic gradient
K = hydraulic conductivity
w = aquifer width

Table 2: Groundwater flux into Frazier Brook determined with seepage meters and piezometers.

	K	i	d	w	Q	Q
	(m/d)	(m/m)	(m)	(m)	(m3/d)	(m3/s)
October 4, 1990	3.67	0.036	823	7.9	860	0.010
November 16, 1990	16.54	0.036	823	7.9	3870	0.045
April 20, 1991	11.27	0.046	823	7.9	3370	0.039
Average					2700	0.031

Table 3: Streamflow results and estimate of groundwater inflow of Frazier Brook along landfill reach.

Location		S5	Walker Brk.	S1	Difference
Streamflow	(m3/s)	0.741	0.457	0.239	0.045
	(ft3/s)	26.17	16.14	8.44	1.59

Table 4: Stream Flow Data.

		Mean Annual Flows		Field	Gaged
	Drainage Area	% of Area	Dingman	4/20/91	4/26/91
	(mi^2)	(cfs)	(cfs)	(cfs)	(cfs)
Blackwater R.	129	214.9*	193	301*	557*
Frazier Brk inlet	21.4	35.7	31.4		50
Walker Brook	10.6	22.8	15.6	16.1	
Frazier Brk above Walker Brook	4.9	10.5	7.2	8.4	

*: From USGS stream gaging station near Webster, NH.

TABLE 5: Metal Concentrations in Groundwater Sampled from Piezometers along Frazier Brook.

	Al (mg/l)	Co (mg/l)	Cr (mg/l)	Cu (mg/l)	Fe (mg/l)	Mg (mg/l)	Mn (mg/l)	Pb (mg/l)	Si (mg/l)	Ti (mg/l)	Zn (mg/l)
P1	11.80	0.000	0.002	0.030	8.98	1.64	0.26	0.041	7.04	0.080	0.096
P2	10.20	0.012	0.000	0.030	37.08	2.13	6.35	0.074	9.24	0.073	0.056
P3	2.43	0.000	0.000	0.020	2.06	1.21	0.16	0.021	4.97	0.028	0.040
P4	1.35	0.130	0.000	0.011	21.40	3.21	6.14	0.000	6.47	0.020	0.028
P5	1.54	0.030	0.000	0.013	84.36	6.48	4.75	0.021	10.37	0.020	0.035
MW-9*			0.001		0.06		0.005	0.005			
GZ-1*			0.001		13.00		2.20	0.005			
EPA Standards**			0.050	1.000	0.30		0.05	0.050			5.000

* Data from December, 1990 (GZA, 1990).

** EPA, 1976

Table 6: Frazier Brook Grab Samples (June,1990). Metal Concentrations to Percent Fines and Combustible Matter Correlations and t-test Results.

Null Hypothesis: There is no correlation between metal concentrations and the amount of fines.

Metal	n	Degrees of Freedom	Corr. Coef. (R)	t-value	t-crit (P=0.05)	Null Hypoth
Al	21	20	0.850	7.027	2.086	Reject
Fe	21	20	0.703	4.306	2.086	Reject
Mg	21	20	0.664	3.868	2.086	Reject
Mn	21	20	0.335	1.548	2.086	Accept
Si	21	20	0.766	5.194	2.086	Reject
As	21	20	0.082	0.361	2.086	Accept
Cd	21	20	0.437	2.118	2.086	Reject
Co	21	20	0.824	6.328	2.086	Reject
Cr	21	20	0.793	5.675	2.086	Reject
Cu	21	20	0.635	3.583	2.086	Reject
Pb	21	20	0.643	3.661	2.086	Reject
Ti	21	20	0.578	3.086	2.086	Reject
Zn	21	20	0.850	7.034	2.086	Reject

Null Hypothesis: There is no correlation between metal concentrations and the amount of combustible matter.

Metal	n	Degrees of Freedom	Corr. Coef. (R)	t-value	t-crit (P=0.05)	Null Hypoth
Al	21	20	0.880	8.080	2.086	Reject
Fe	21	20	0.751	4.954	2.086	Reject
Mg	21	20	0.679	4.031	2.086	Reject
Mn	21	20	0.495	2.484	2.086	Reject
Si	21	20	0.819	6.213	2.086	Reject
As	21	20	0.073	0.319	2.086	Accept
Cd	21	20	0.469	2.313	2.086	Reject
Co	21	20	0.816	6.149	2.086	Reject
Cr	21	20	0.783	5.493	2.086	Reject
Cu	21	20	0.634	3.571	2.086	Reject
Pb	21	20	0.744	4.860	2.086	Reject
Ti	21	20	0.551	2.878	2.086	Reject
Zn	21	20	0.857	7.235	2.086	Reject

Table 7: Frazier Brook Grab Samples (June,1990). Metal Concentrations to Percent Fines plus Combustible Matter Multiple Correlation and F-test Results.

Null Hypothesis: There is no correlation between metal concentrations and the amount of fines plus combustible matter.

Metal	Multiple Corr. Coef. (R)	(R^2)	F-value	F-crit (P=0.05)	Null Hypoth.
Al	0.887	0.787	33.287	3.550	Reject
Fe	0.752	0.566	11.707	3.550	Reject
Mg	0.687	0.472	8.056	3.550	Reject
Mn	0.574	0.329	4.415	3.550	Reject
Si	0.820	0.672	18.448	3.550	Reject
As	0.083	0.007	0.062	3.550	Accept
Cd	0.469	0.220	2.541	3.550	Accept
Co	0.838	0.702	21.250	3.550	Reject
Cr	0.806	0.650	16.701	3.550	Reject
Cu	0.649	0.421	6.532	3.550	Reject
Pb	0.750	0.563	11.571	3.550	Reject
Ti	0.581	0.338	4.577	3.550	Reject
Zn	0.872	0.760	28.685	3.550	Reject

Table 8: Mann-Whitney and Wilcoxon statistical test results for comparing metal concentration changes in Frazier Brook sediments.

Null Hypothesis: There is no difference in the metal concentrations in the two time periods (June & October, 1990).

MANN-WHITNEY

Metal	Z (+ -)	Z crit	Null Hypoth.	Z crit	Null Hypoth.
		P=0.10		P=0.05	
Al	1.85	1.64	Reject	1.96	Accept
Fe	2.53	1.64	Reject	1.96	Reject
Mg	1.97	1.64	Reject	1.96	Reject
Mn	1.87	1.64	Reject	1.96	Accept
Si	1.17	1.64	Accept	1.96	Accept
As					
Cd	3.48	1.64	Reject	1.96	Reject
Co	2.35	1.64	Reject	1.96	Reject
Cr	2.25	1.64	Reject	1.96	Reject
Cu	0.73	1.64	Accept	1.96	Accept
Pb	1.32	1.64	Accept	1.96	Accept
Ti	2.25	1.64	Reject	1.96	Reject
Zn	4.31	1.64	Reject	1.96	Reject

WILCOXON

Metal	Z (+ -)	Z crit	Null Hypoth.	Z crit	Null Hypoth.
		P=0.10		P=0.05	
Al	2.31	1.64	Reject	1.96	Reject
Fe	2.97	1.64	Reject	1.96	Reject
Mg	1.96	1.64	Reject	1.96	Reject
Mn	2.76	1.64	Reject	1.96	Reject
Si	1.30	1.64	Accept	1.96	Accept
As					
Cd	3.22	1.64	Reject	1.96	Reject
Co	3.28	1.64	Reject	1.96	Reject
Cr	2.59	1.64	Reject	1.96	Reject
Cu	0.64	1.64	Accept	1.96	Accept
Pb	0.82	1.64	Accept	1.96	Accept
Ti	2.59	1.64	Reject	1.96	Reject
Zn	4.01	1.64	Reject	1.96	Reject

Table 9: Eagle Pond Grab Samples (June,1990). Metal Concentrations to Percent Fines and Combustible Matter Correlations and t-test Results.

Null Hypothesis: There is no correlation between metal concentrations and the amount of fines.

Metal	n	Degrees of Freedom	Corr. Coef (R)	t-value	t-crit (P=0.05)	Null Hypoth
Al	19	18	0.911	9.081	2.101	Reject
Fe	19	18	0.685	3.877	2.101	Reject
Mg	19	18	0.683	3.856	2.101	Reject
Mn	19	18	0.165	0.690	2.101	Accept
Si	19	18	0.732	4.429	2.101	Reject
As	19	18	0.059	0.243	2.101	Accept
Cd	19	18	0.755	4.754	2.101	Reject
Co	19	18	0.461	2.145	2.101	Reject
Cr	18	17	0.923	9.577	2.110	Reject
Cu	19	18	0.698	4.023	2.101	Reject
Pb	19	18	0.838	6.333	2.101	Reject
Ti	18	17	0.452	2.026	2.110	Accept
Zn	19	18	0.781	5.163	2.101	Reject

Null Hypothesis: There is no correlation between metal concentrations and the amount of combustible matter.

Metal	n	Degrees of Freedom	Corr. Coef (R)	t-value	t-crit (P=0.05)	Null Hypoth
Al	19	18	0.783	5.191	2.101	Reject
Fe	19	18	0.535	2.612	2.101	Reject
Mg	19	18	0.396	1.780	2.101	Accept
Mn	19	18	0.623	3.288	2.101	Reject
Si	19	18	0.474	2.221	2.101	Reject
As	19	18	0.003	0.011	2.101	Accept
Cd	19	18	0.845	6.516	2.101	Reject
Co	19	18	0.716	4.223	2.101	Reject
Cr	18	17	0.757	4.634	2.110	Reject
Cu	19	18	0.731	4.410	2.101	Reject
Pb	19	18	0.805	5.597	2.101	Reject
Ti	18	17	0.178	0.725	2.110	Accept
Zn	19	18	0.835	6.265	2.101	Reject

Table 10: Eagle Pond Grab Samples (June,1990). Metal Concentrations to Percent Fines plus Combustible Matter Multiple Correlation and F-test Results.

Null Hypothesis: There is no correlation between metal concentrations and the amount of fines plus combustible matter.

Metal	Multiple Corr. Coef. (R)	(R^2)	F-value	F-crit (P=0.05)	Null Hypoth
Al	0.920	0.846	43.828	3.630	Reject
Fe	0.685	0.469	7.079	3.630	Reject
Mg	0.712	0.507	8.246	3.630	Reject
Mn	0.795	0.632	13.746	3.630	Reject
Si	0.745	0.555	9.988	3.630	Reject
As	0.089	0.008	0.064	3.630	Accept
Cd	0.861	0.741	22.905	3.630	Reject
Co	0.729	0.531	9.077	3.630	Reject
Cr	0.924	0.854	43.634	3.680	Reject
Cu	0.761	0.579	11.004	3.630	Reject
Pb	0.875	0.766	26.064	3.630	Reject
Ti	0.539	0.291	3.072	3.680	Accept
Zn	0.863	0.745	23.342	3.630	Reject

Table 11: Mann-Whitney statistical test results for comparing metal concentration changes in Eagle Pond bottom sediments.

Null Hypothesis: There is no difference in the metal concentrations in the two time periods (June & October, 1990).

MANN-WHITNEY

Metal	Z (+ -)	Z crit	Null Hypoth.	Z crit	Null Hypoth.
		P=0.10		P=0.05	
Al	1.52	1.64	Accept	1.96	Accept
Fe	2.30	1.64	Reject	1.96	Reject
Mg	2.25	1.64	Reject	1.96	Reject
Mn	1.10	1.64	Accept	1.96	Accept
Si	1.35	1.64	Accept	1.96	Accept
As					
Cd	1.85	1.64	Reject	1.96	Accept
Co	1.85	1.64	Reject	1.96	Accept
Cr	1.90	1.64	Reject	1.96	Accept
Cu	0.31	1.64	Accept	1.96	Accept
Pb	0.84	1.64	Accept	1.96	Accept
Ti	2.72	1.64	Reject	1.96	Reject
Zn	1.97	1.64	Reject	1.96	Reject

Table 12: Core A (June, 1990). Metal Concentrations to Percent Fines and Combustible Matter Correlations and t-test Results.

Null Hypothesis: There is no correlation between metal concentrations and the amount of fines.

Metal	n	Degrees of Freedom	Corr. Coef. (R)	t-value	t-crit (P=0.05)	Null Hypoth
Al	22	21	0.735	4.846	2.080	Reject
Fe	22	21	0.671	4.047	2.080	Reject
Mg	22	21	0.593	3.292	2.080	Reject
Mn	22	21	0.777	5.526	2.080	Reject
Si	22	21	0.773	5.444	2.080	Reject
As	22	21	0.229	1.051	2.080	Accept
Cd	22	21	0.655	3.874	2.080	Reject
Co	22	21	0.759	5.206	2.080	Reject
Cr	22	21	0.722	4.668	2.080	Reject
Cu	22	21	0.588	3.253	2.080	Reject
Pb	22	21	0.651	3.832	2.080	Reject
Ti	21	20	0.261	1.177	2.086	Accept
Zn	22	21	0.752	5.110	2.080	Reject

Null Hypothesis: There is no correlation between metal concentrations and the amount of combustible matter.

Metal	n	Degrees of Freedom	Corr. Coef. (R)	t-value	t-crit (P=0.05)	Null Hypoth
Al	22	21	0.512	2.665	2.080	Reject
Fe	22	21	0.531	2.804	2.080	Reject
Mg	22	21	0.541	2.875	2.080	Reject
Mn	22	21	0.598	3.338	2.080	Reject
Si	22	21	0.577	3.158	2.080	Reject
As	22	21	0.071	0.318	2.080	Accept
Cd	22	21	0.495	2.545	2.080	Reject
Co	22	21	0.555	2.984	2.080	Reject
Cr	22	21	0.517	2.699	2.080	Reject
Cu	22	21	0.580	3.181	2.080	Reject
Pb	22	21	0.556	2.995	2.080	Reject
Ti	21	20	0.391	1.851	2.086	Accept
Zn	22	21	0.592	3.289	2.080	Reject

Table 13: Core A (June, 1990). Metal Concentrations to Percent Fines plus Combustible Matter Multiple Correlations and F-test Results

Null Hypothesis: There is no correlation between metal concentrations and th
amount of fines and combustible matter.

Metal	Multiple Corr. Coef. (R)	(R^2)	F-value	F-crit (P=0.05)	Null Hypoth.
Al	0.749	0.561	12.137	3.520	Reject
Fe	0.683	0.466	8.302	3.520	Reject
Mg	0.640	0.410	6.596	3.520	Reject
Mn	0.794	0.630	16.177	3.520	Reject
Si	0.781	0.610	14.840	3.520	Reject
As	0.224	0.050	0.502	3.520	Accept
Cd	0.678	0.460	8.062	3.520	Reject
Co	0.772	0.596	13.970	3.520	Reject
Cr	0.734	0.539	11.089	3.520	Reject
Cu	0.670	0.449	7.731	3.520	Reject
Pb	0.696	0.484	8.907	3.520	Reject
Ti	0.391	0.153	1.629	3.550	Accept
Zn	0.778	0.605	14.584	3.520	Reject

Table 14: Metal Concentrations to Aluminum Correlation and t-test Results for Core B Background Sections.

Null Hypothesis: There is no correlation between the concentrations of metals and aluminum.

Metal	n	Degrees of Freedom	Corr. Coef.	t-value	t-crit (P=0.05)	Null Hypoth
Fe	18	17	0.793	5.208	2.110	Reject
Mg	18	17	0.922	9.502	2.110	Reject
Mn	18	17	0.698	3.895	2.110	Reject
Si	18	17	0.978	18.837	2.110	Reject
As	18	17	0.193	0.786	2.110	Accept
Cd	18	17	0.075	0.300	2.110	Accept
Co	18	17	0.685	3.757	2.110	Reject
Cr	18	17	0.779	4.977	2.110	Reject
Cu	18	17	0.860	6.740	2.110	Reject
Pb	18	17	0.406	1.778	2.110	Accept
Ti	18	17	0.833	6.028	2.110	Reject
Zn	18	17	0.588	2.911	2.110	Reject

Table 15:	Core C: Water Content by Mass and by Volume (porosity)					
						Cumulative
	Uncompacte	Water			Normalized	Compacted
Sample #	Depth	Content	Porosity	Wet:Dry	Wet:Dry	Depth
	(cm)	%	%	Vol Ratio	Ratio	(cm)
EP:CC-1	1	81.51	92.12	12.69	0.50	0.50
EP:CC-2	2	82.14	92.43	13.21	0.48	0.98
EP:CC-3	3	81.68	92.21	12.84	0.50	1.48
EP:CC-4	4	82.29	92.51	13.34	0.48	1.96
EP:CC-5	5	79.45	91.12	11.26	0.57	2.53
EP:CC-6	6	82.13	92.42	13.19	0.49	3.02
EP:CC-7	7	82.07	92.37	13.11	0.49	3.50
EP:CC-8	8	82.67	92.69	13.67	0.47	3.97
EP:CC-9	9	81.75	92.24	12.88	0.50	4.47
EP:CC-10	10	81.27	91.98	12.48	0.51	4.98
EP:CC-11	11	80.43	91.60	11.90	0.54	5.52
EP:CC-12	12	80.64	91.70	12.05	0.53	6.05
EP:CC-13	13	80.23	91.50	11.77	0.54	6.59
EP:CC-14	14	78.40	90.57	10.61	0.60	7.20
EP:CC-15	15	77.96	90.35	10.37	0.62	7.82
EP:CC-16	16	76.49	89.60	9.62	0.67	8.48
EP:CC-17	17	75.69	89.20	9.26	0.69	9.17
EP:CC-18	18	73.96	88.27	8.52	0.75	9.92
EP:CC-19	19	72.63	87.55	8.03	0.80	10.72
EP:CC-20	20	69.57	85.83	7.06	0.91	11.63
EP:CC-21	21	69.40	85.75	7.02	0.91	12.54
EP:CC-22	22	68.79	85.39	6.85	0.93	13.47
EP:CC-23	23	71.36	86.85	7.60	0.84	14.32
EP:CC-24	24	74.51	88.56	8.74	0.73	15.05
EP:CC-25	26	74.51	88.57	8.75	0.73	15.78
EP:CC-26	28	73.81	88.20	8.47	0.76	16.53
EP:CC-27	30	74.75	88.70	8.85	0.72	17.26
EP:CC-28	32	75.34	89.01	9.10	0.70	17.96
EP:CC-29	34	74.69	88.66	8.82	0.73	18.69
EP:CC-30	36	73.68	88.12	8.42	0.76	19.45
EP:CC-31	38	74.21	88.40	8.62	0.74	20.19
EP:CC-32	40	73.92	88.24	8.51	0.75	20.94
EP:CC-33	43	71.63	86.99	7.69	0.83	21.77
EP:CC-34	46	69.33	85.69	6.99	0.92	22.69
EP:CC-35	49	68.85	85.42	6.86	0.93	23.62
EP:CC-36	52	67.09	84.38	6.40	1.00	24.62

LIST OF FIGURES:

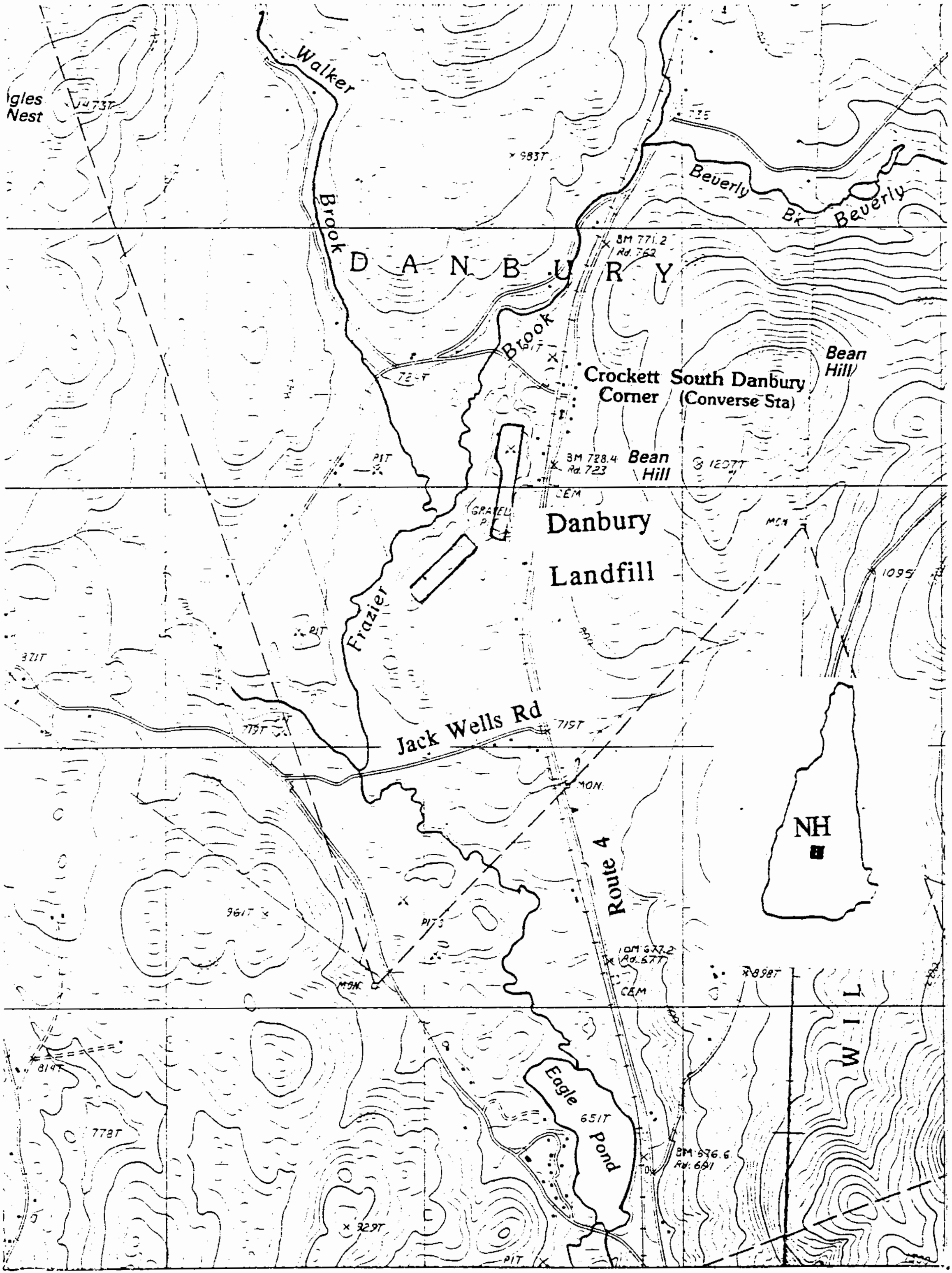


Figure 1: Topographic Site Map. Danbury Landfill, Frazier Brook, and Eagle Pond.

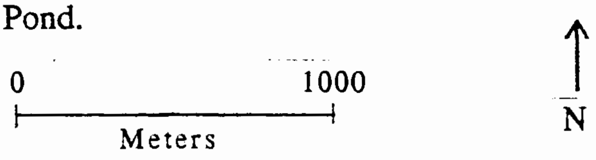
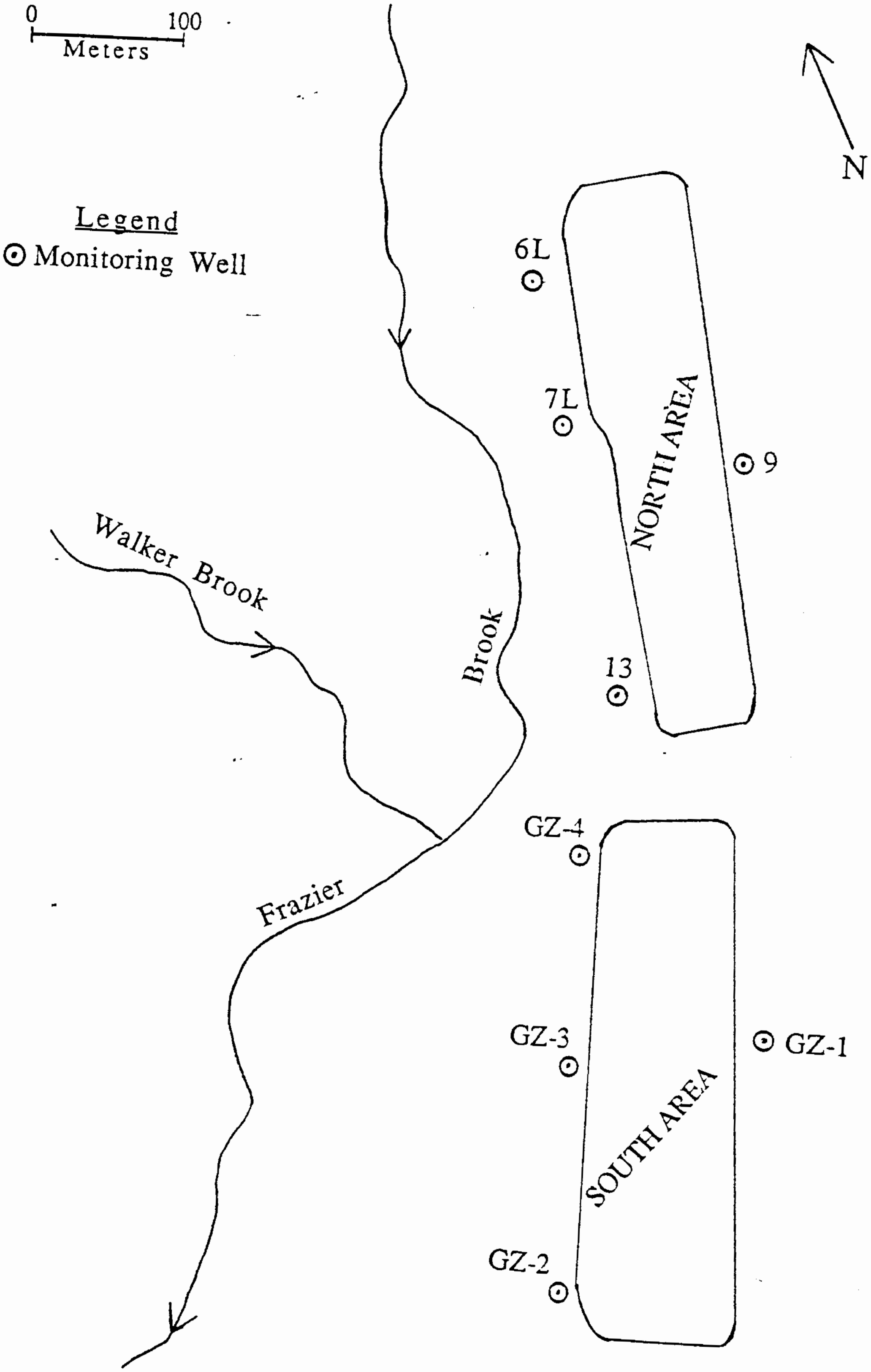
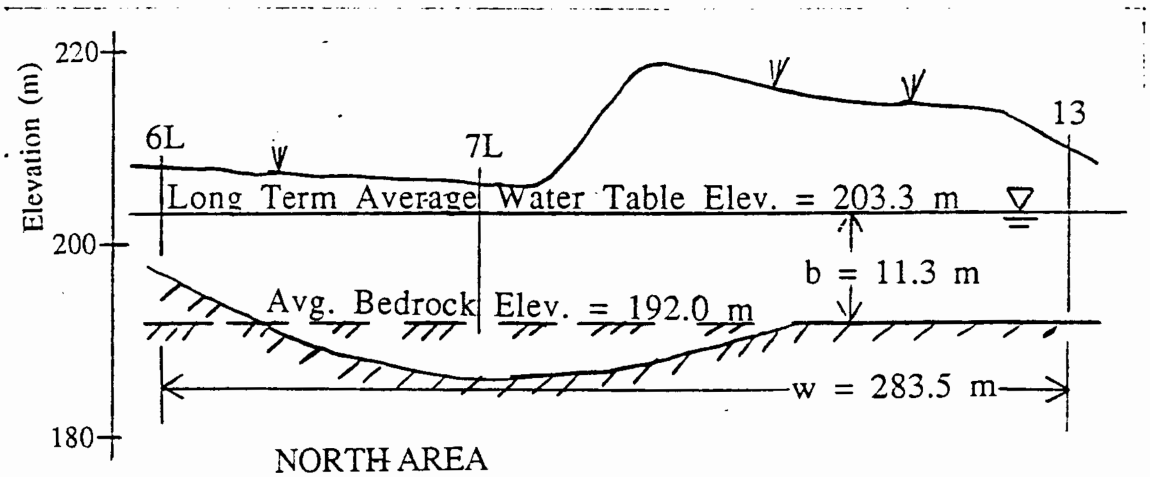


Figure 2: Landfill Site Map (from GZA, 1989).





NORTH AREA

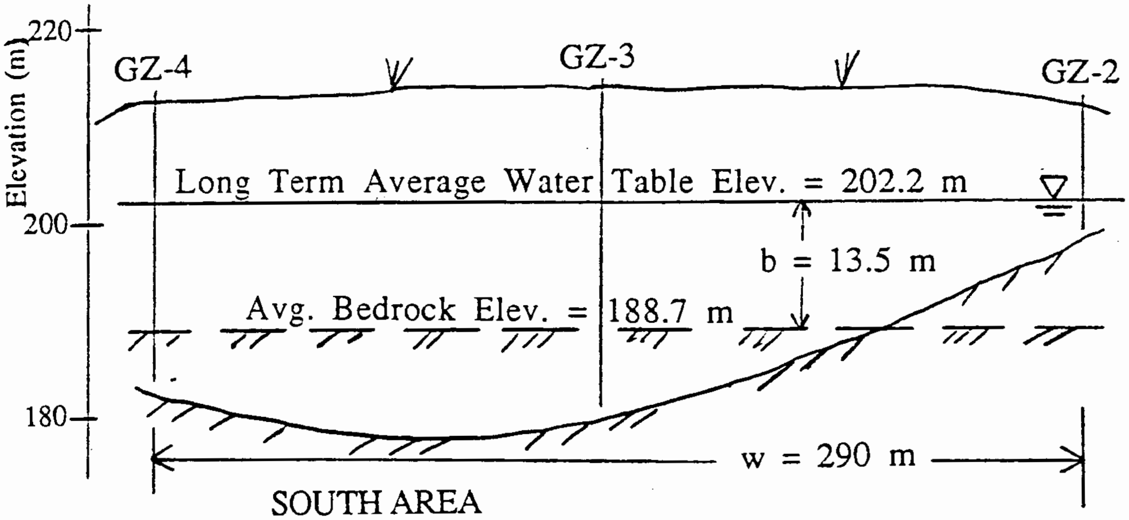
K = 15.2 m/d

i = 0.025

b = 11.3 m

w = 283.5 m

$Q = Kibw = 1217\ m^3/d$
$= 0.014\ m^3/s$



SOUTH AREA

K = 15.2 m/d

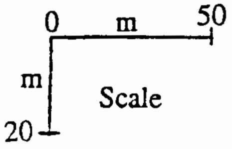
i = 0.018

b = 13.5 m

w = 290 m

$Q = Kibw = 1071\ m^3/d$
$= 0.012\ m^3/s$

Figure 3: Cross Sections of Landfill Areas *
and Groundwater Flow Calculations.



* from Gallup (1982); and GZA (1989)

Figure 4: Seepage Meter, Piezometer and Stream Gaging Locations.

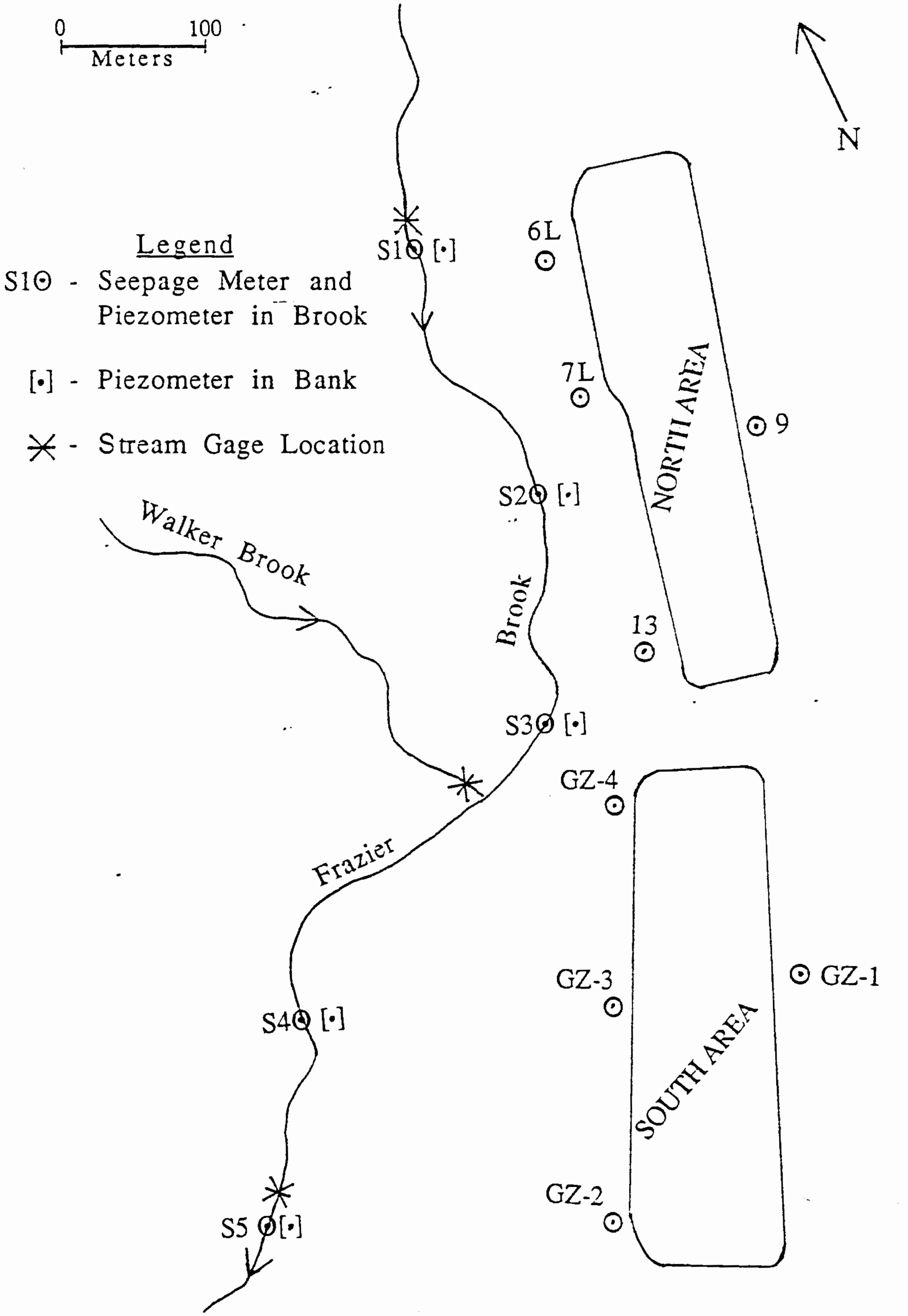
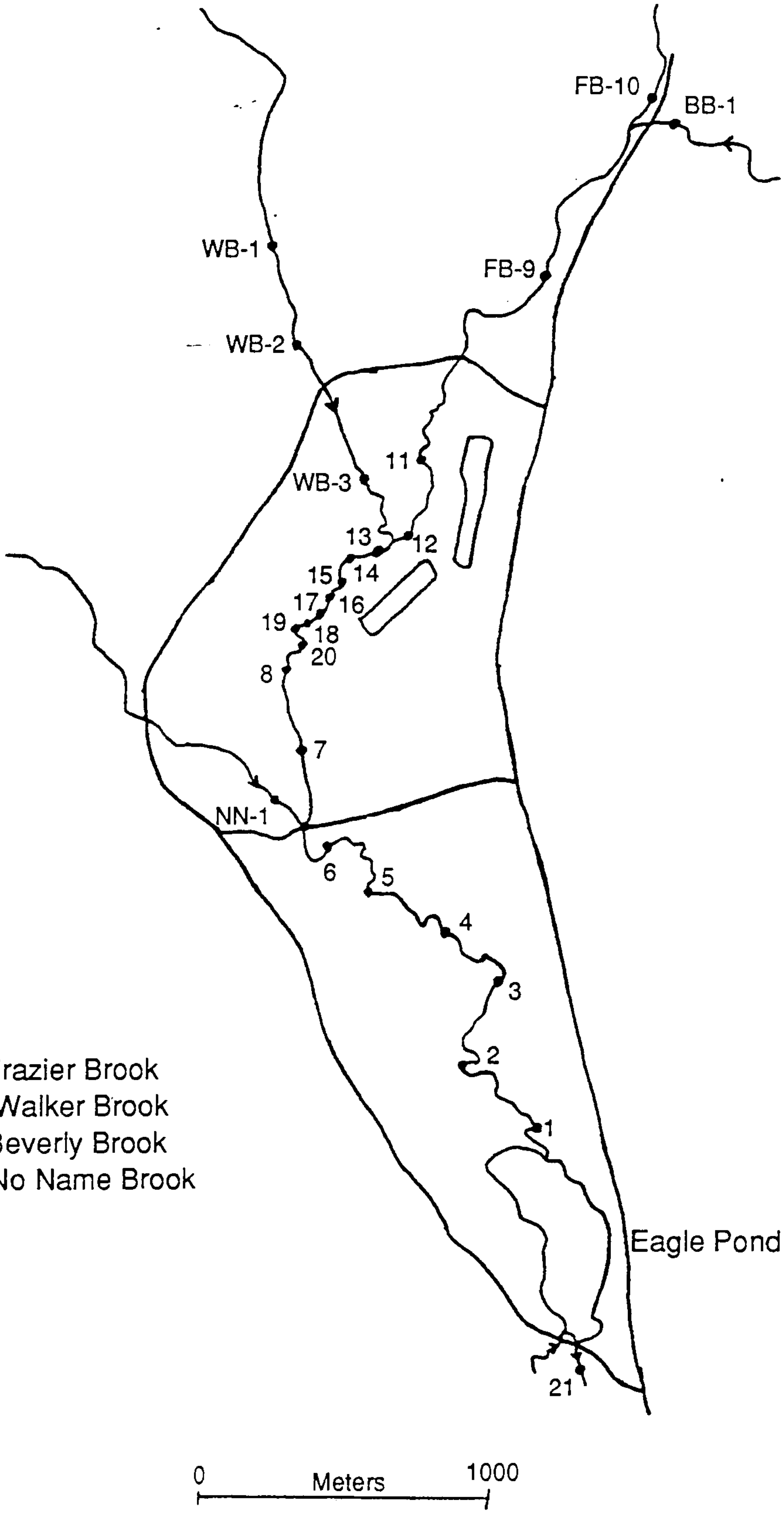
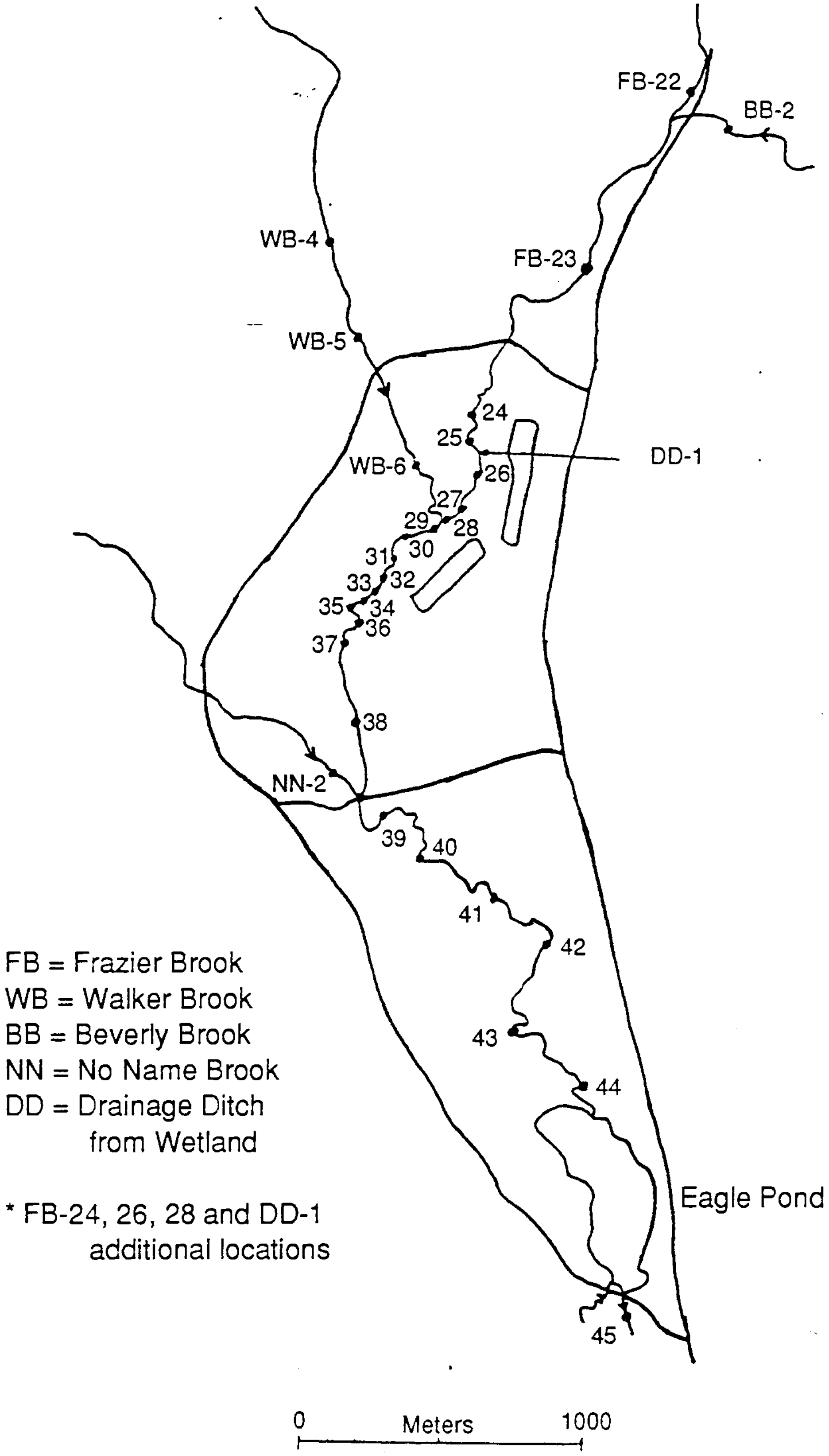


Figure 5a: Sediment Sampling Locations (June, 1990).



FB = Frazier Brook
WB = Walker Brook
BB = Beverly Brook
NN = No Name Brook

Figure 5b: Sediment Sampling Locations (October, 1990).



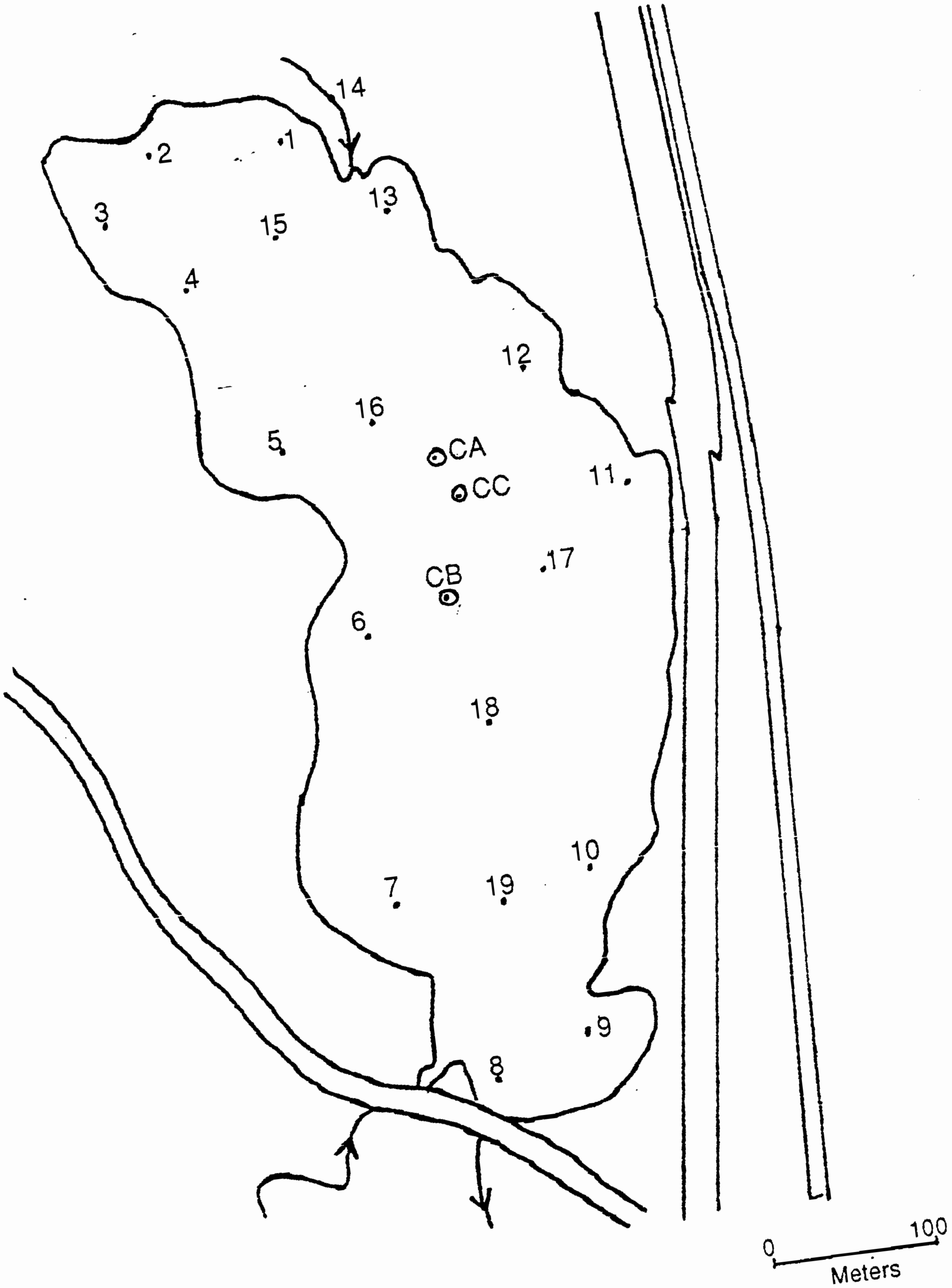


Figure 6a: Sediment Sampling Locations in Eagle Pond (June, 1990).

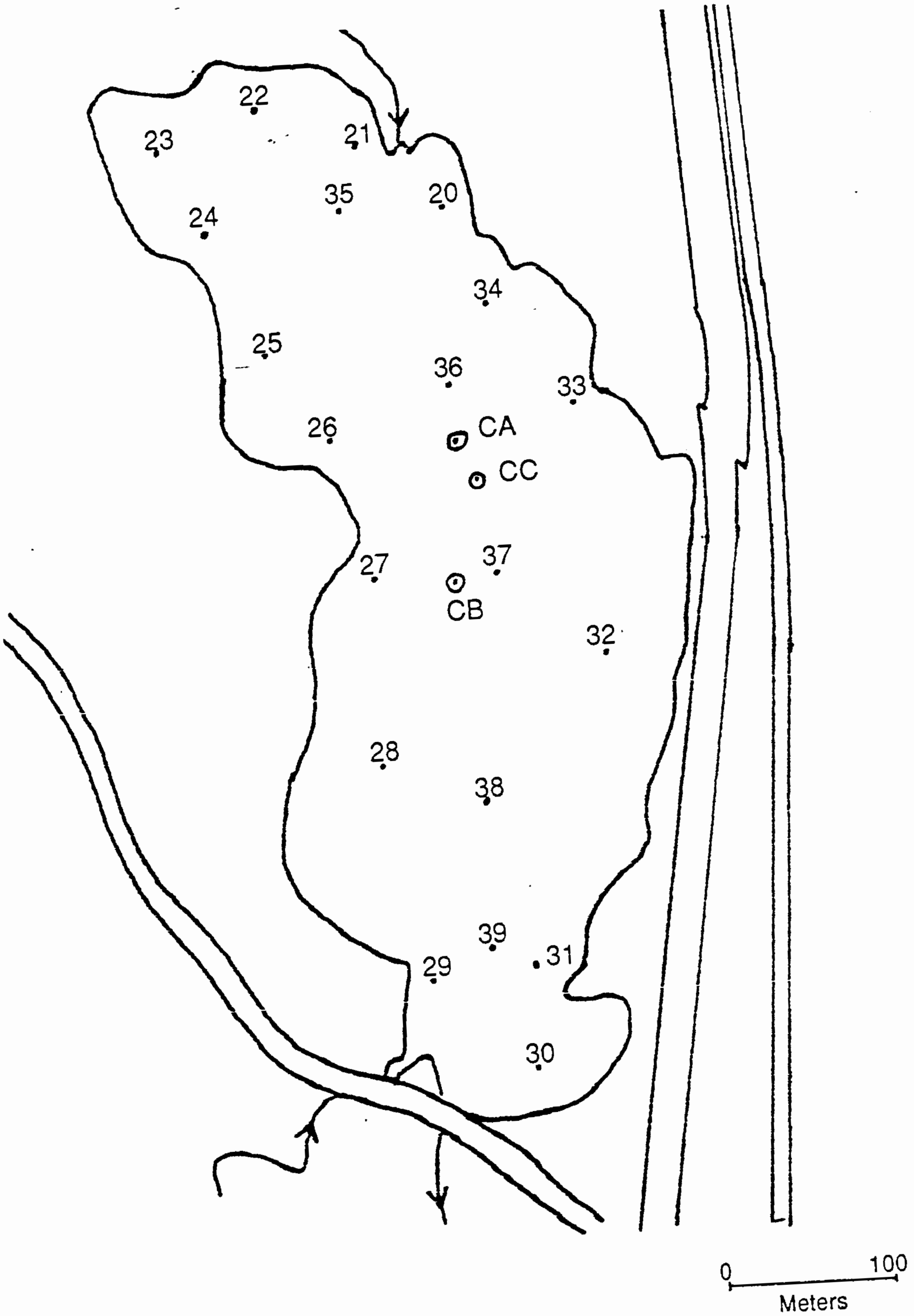
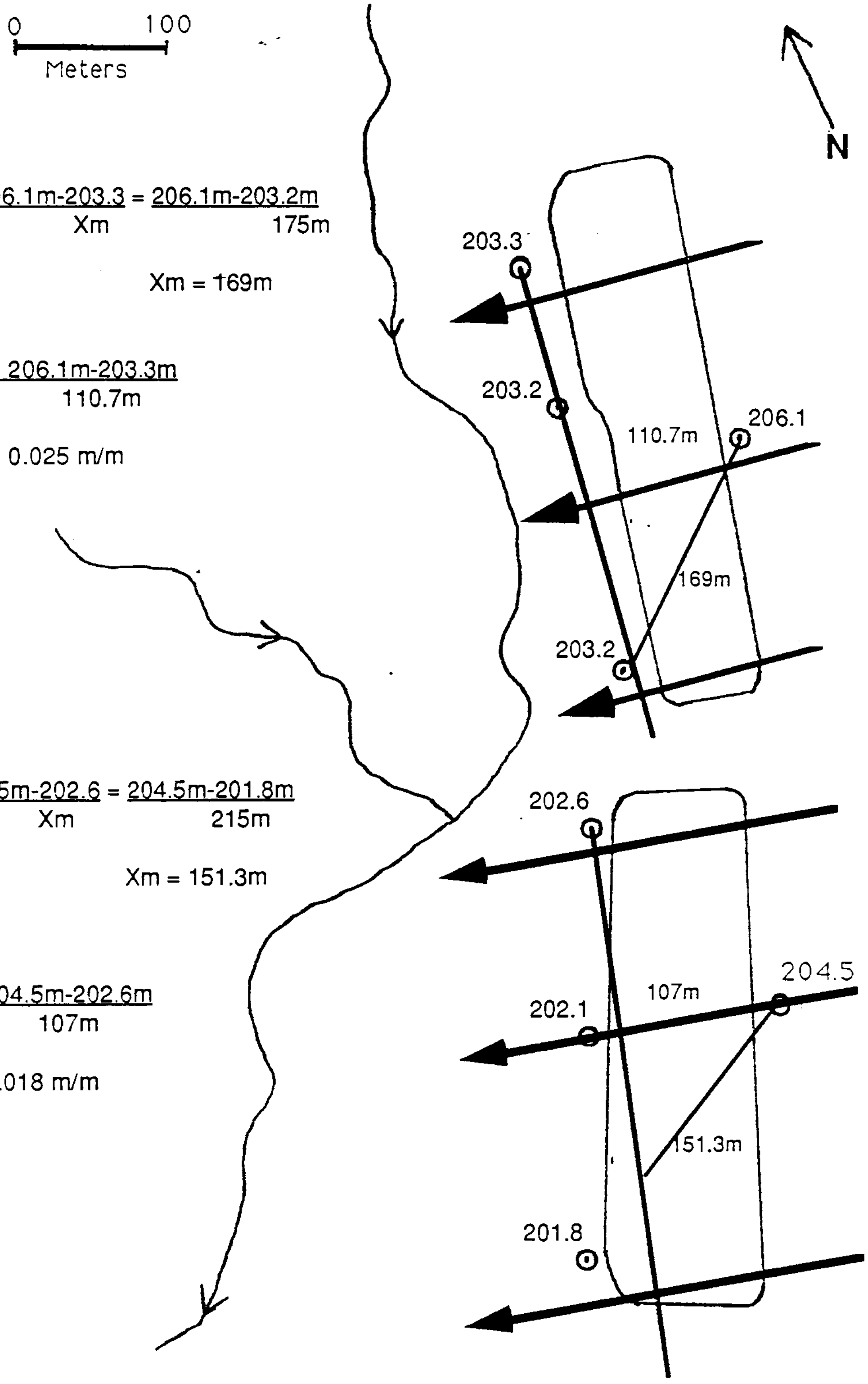


Figure 6b: Sediment Sampling Locations, Eagle Pond (October, 1990).

Figure 7: Long Term Average Groundwater Flow Direction and Hydraulic Gradient (i).



$$\frac{206.1m-203.3}{Xm} = \frac{206.1m-203.2m}{175m}$$

$Xm = 169m$

$$i = \frac{206.1m-203.3m}{110.7m}$$

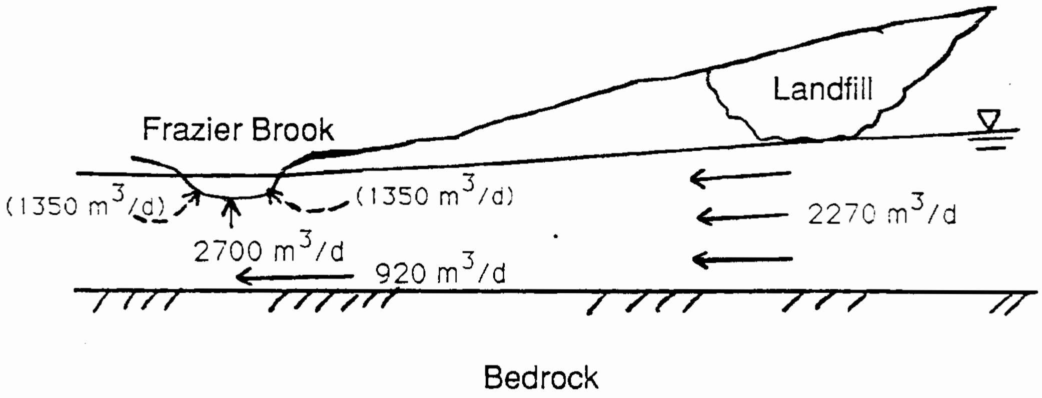
$i = 0.025$ m/m

$$\frac{204.5m-202.6}{Xm} = \frac{204.5m-201.8m}{215m}$$

$Xm = 151.3m$

$$i = \frac{204.5m-202.6m}{107m}$$

$i = 0.018$ m/m



Not to Scale

Figure 8: Average Groundwater Flow Rates Through Landfill and into Frazier Brook.

Approximately 2700 m^3/d of groundwater enters Frazier Brook (estimated with seepage meters and piezometers). If assume both sides contribute 1/2 of this total then flow through landfill contributes 1350 m^3/d to brook and 920 m^3/d to regional groundwater system.

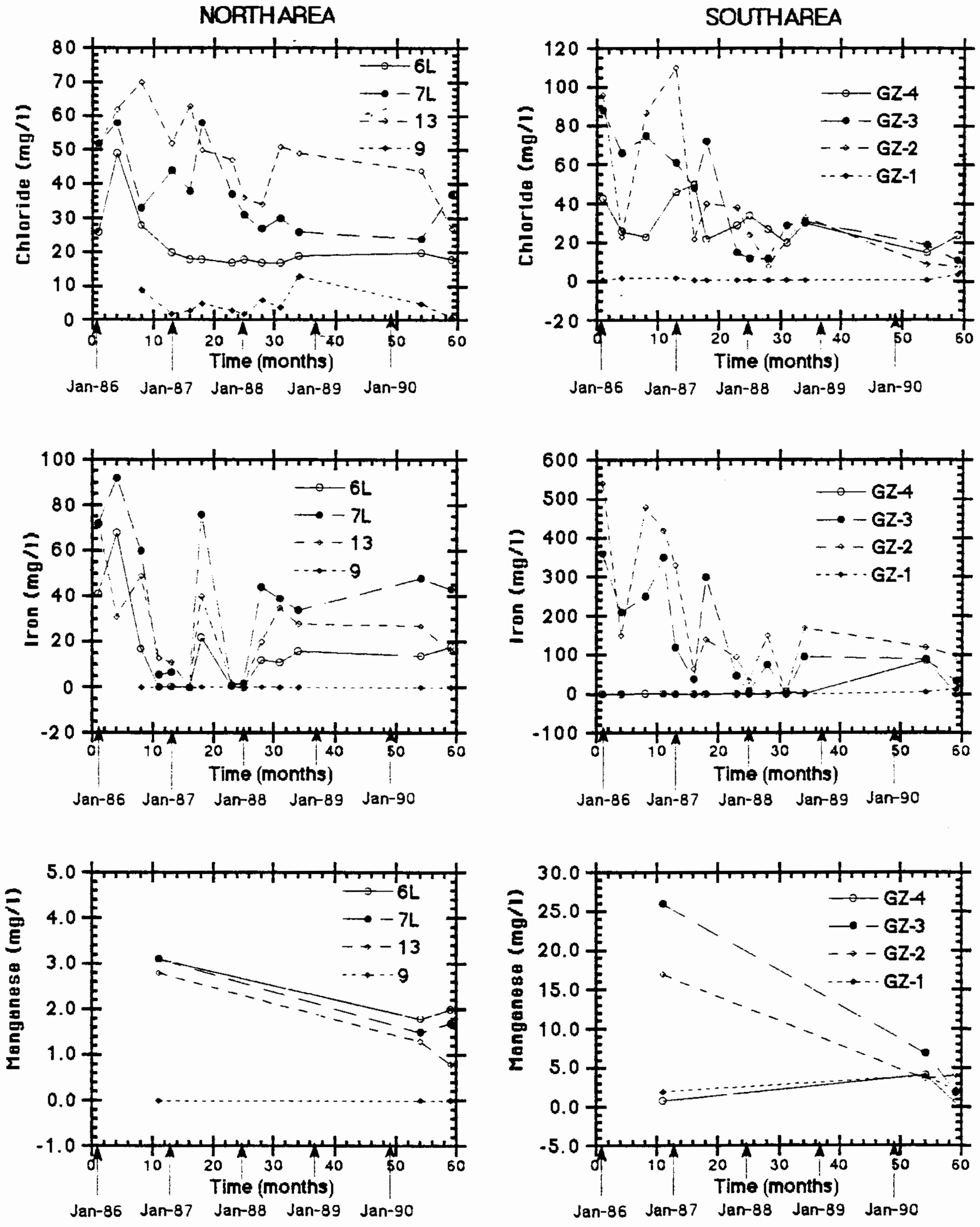


Figure 9: Landfill Monitoring Wells, Groundwater Quality over Time.

Figure 10: Geophysical Survey Lines

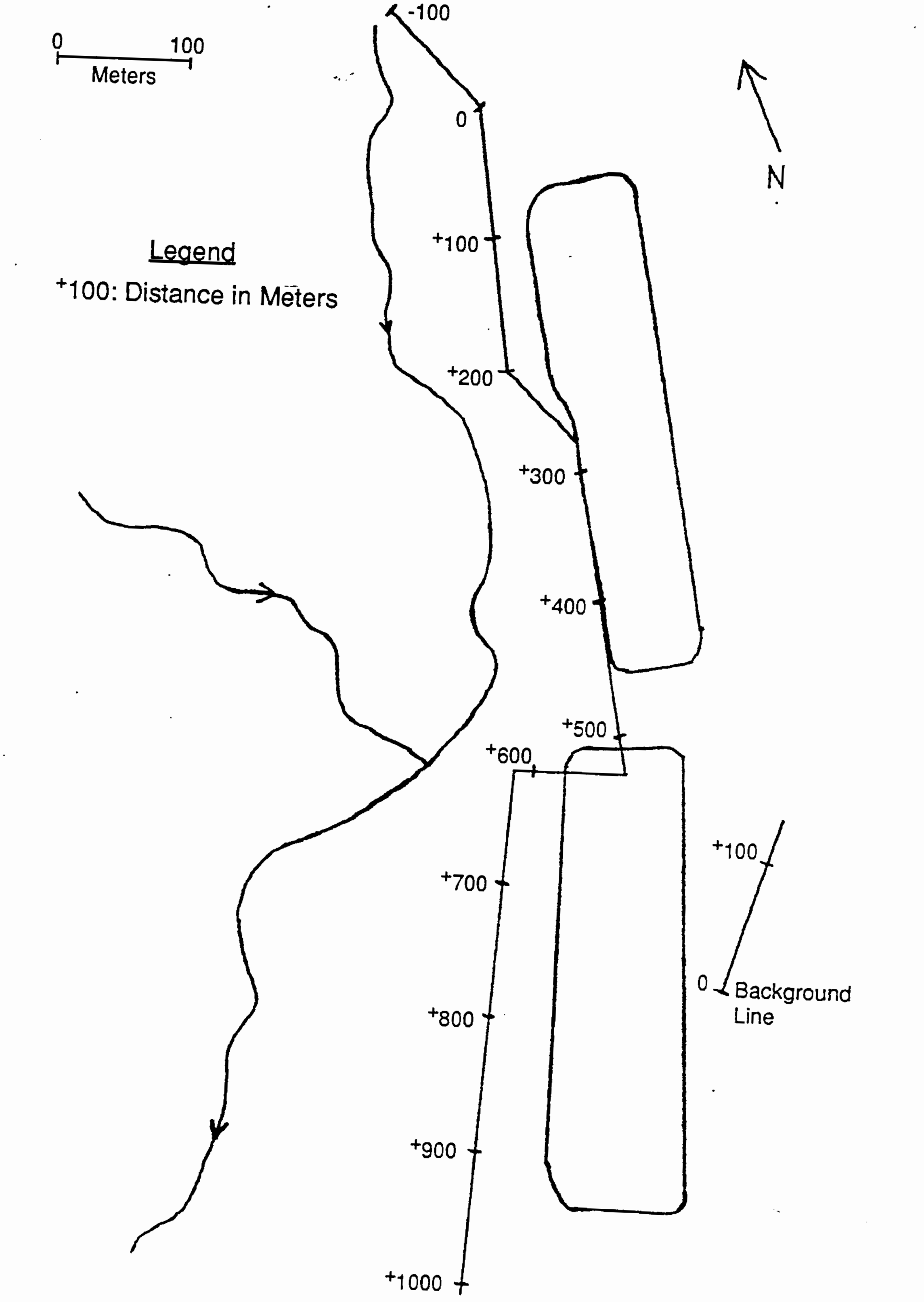


Figure 11a.

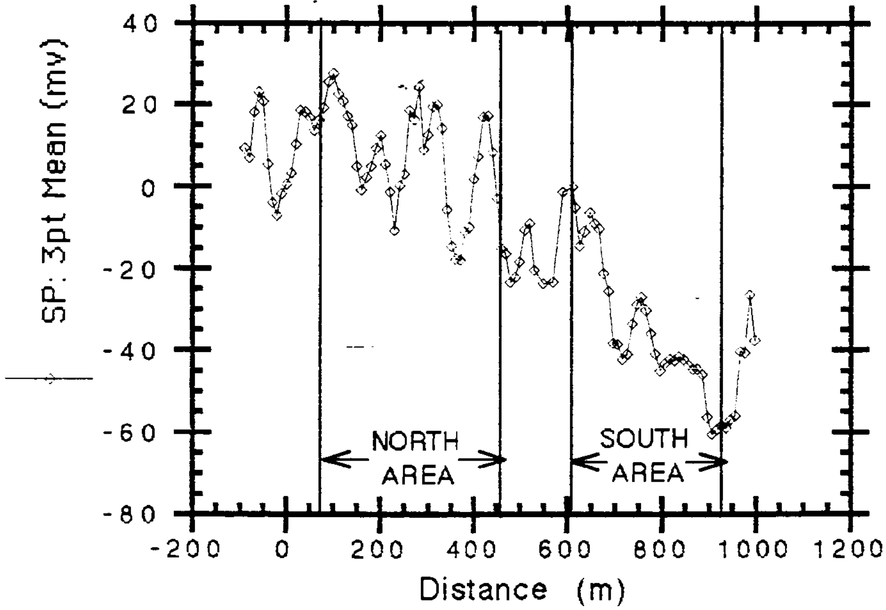


Figure 11b.

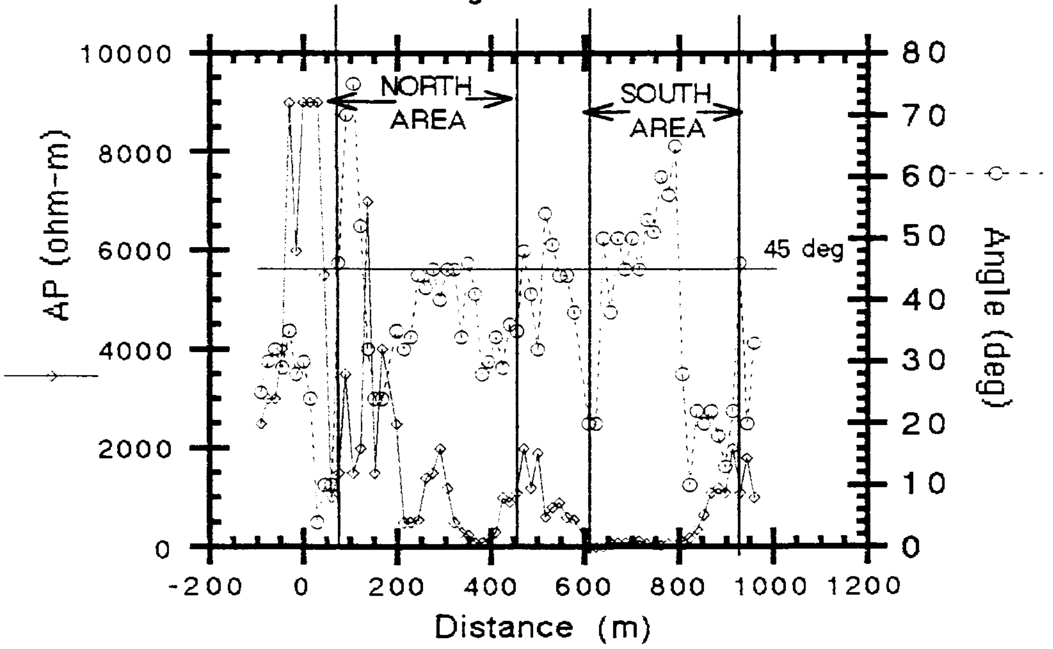


Figure 11a: Spontaneous Potential (smoothed with 3 point running mean) vs Distance along Landfill.
11b: Apparent Resistivity (AP) and Phase Angle vs Distance.

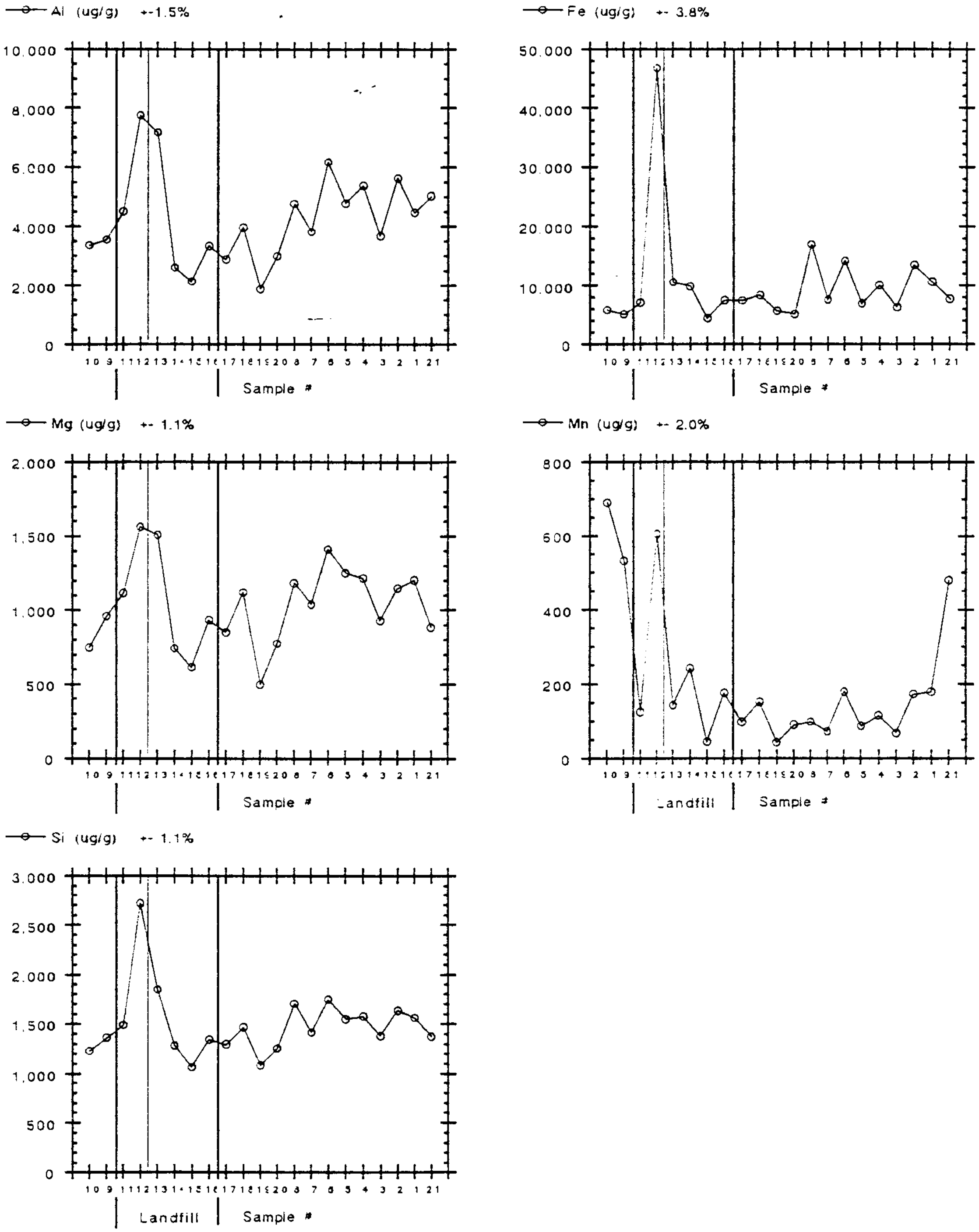


Figure 12: Frazier Brook Grab Samples (June, 1990). Major Metal Concentration per Gram Sediment.

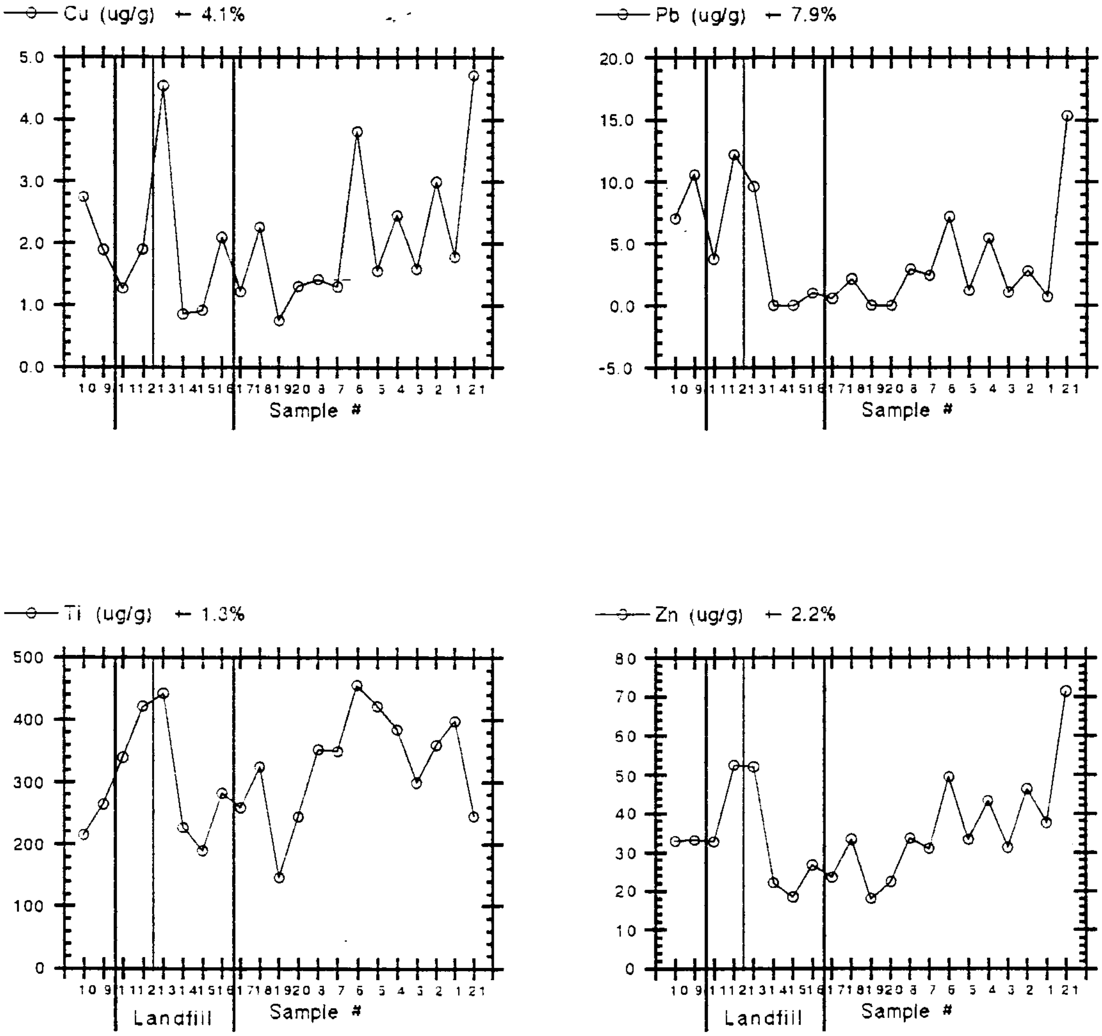


Figure 12: Frazier Brook Grab Samples (June, 1990). Trace Metal Concentration per Gram Sediment.

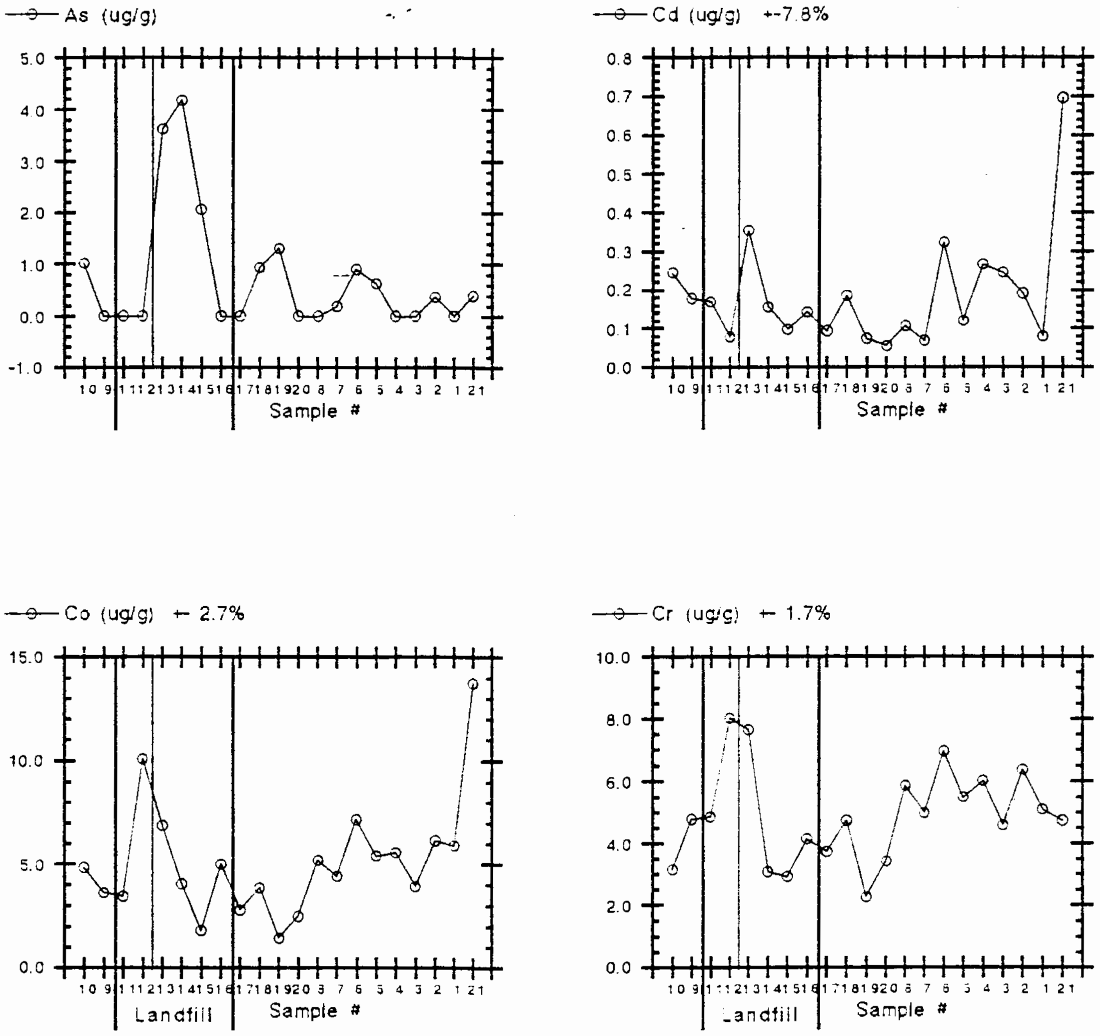


Figure 12: Frazier Brook Grab Samples (June, 1990). Trace Metal Concentration per Gram Sediment.

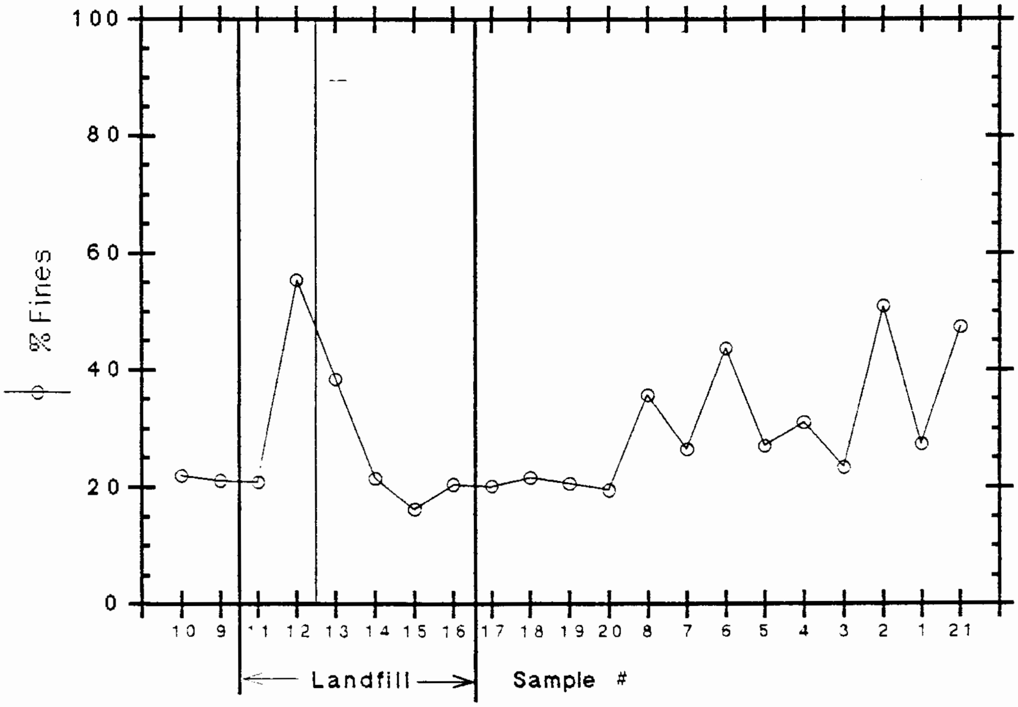


Figure 13: Distribution of Fine Grains in Frazier Brook Sediments.

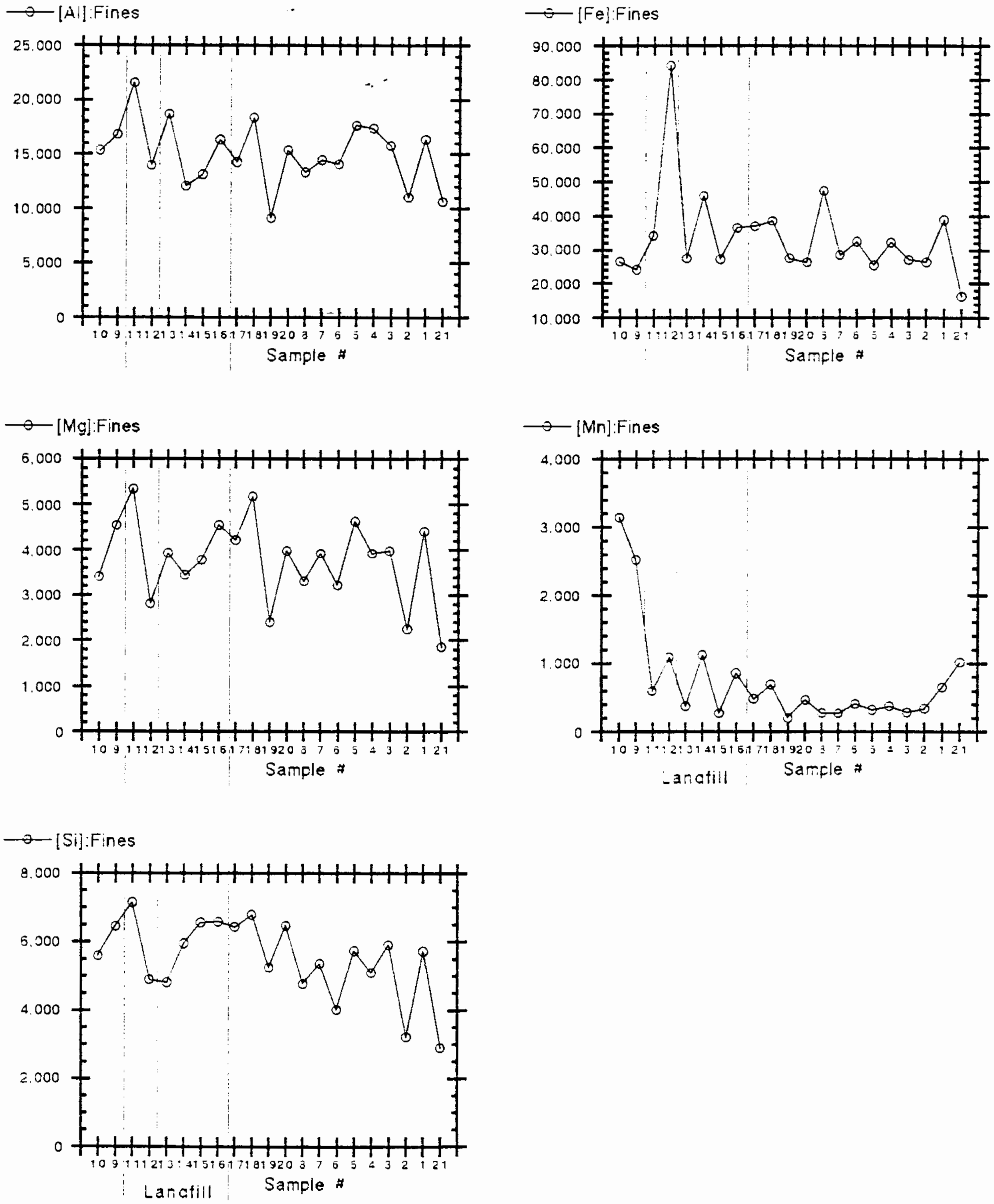


Figure 14: Frazier Brook Grab Samples (June,1990). Major Metals to Percent Fines Normalized Values.

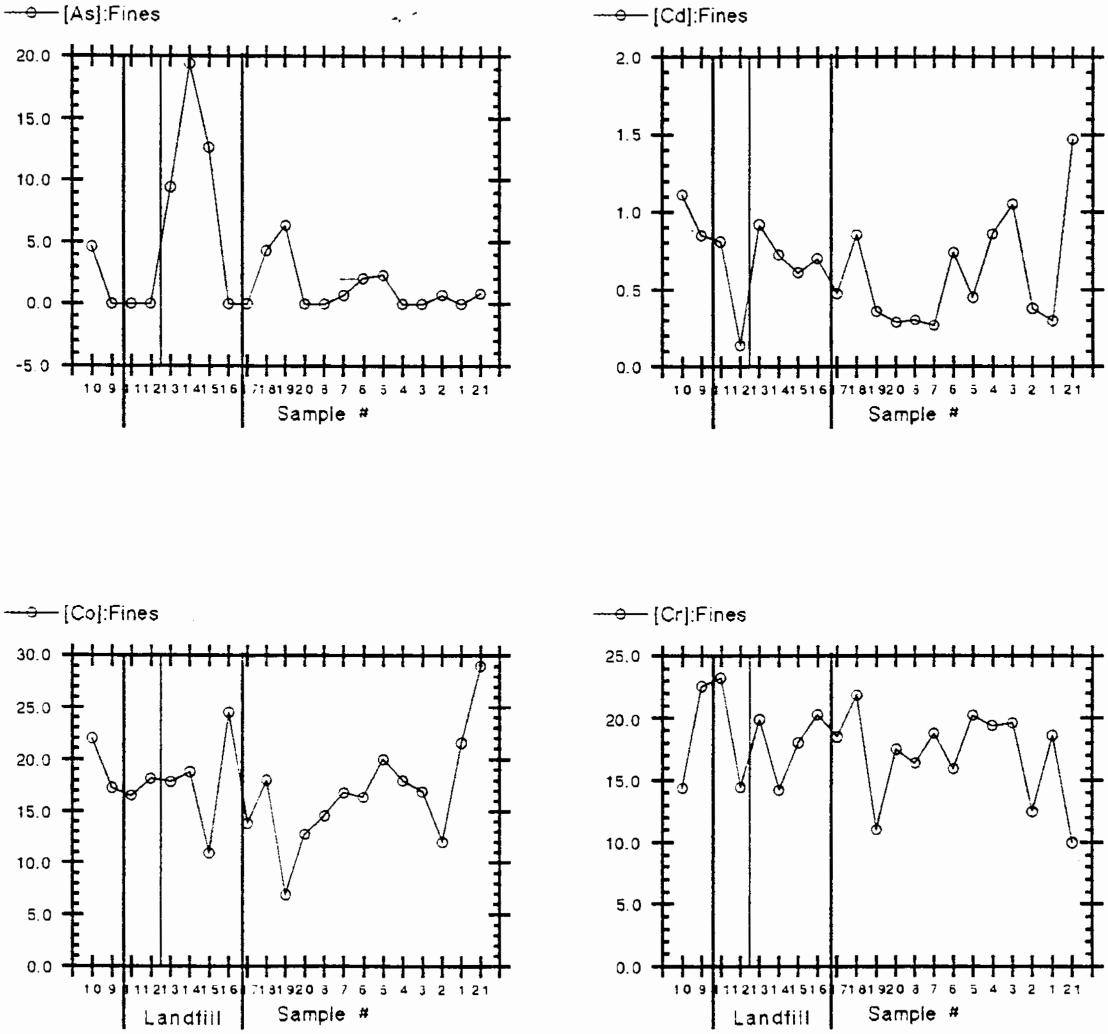


Figure 14: Frazier Brook Grab Samples (June, 1990).
Trace Metals to Percent Fines Normalized Values.

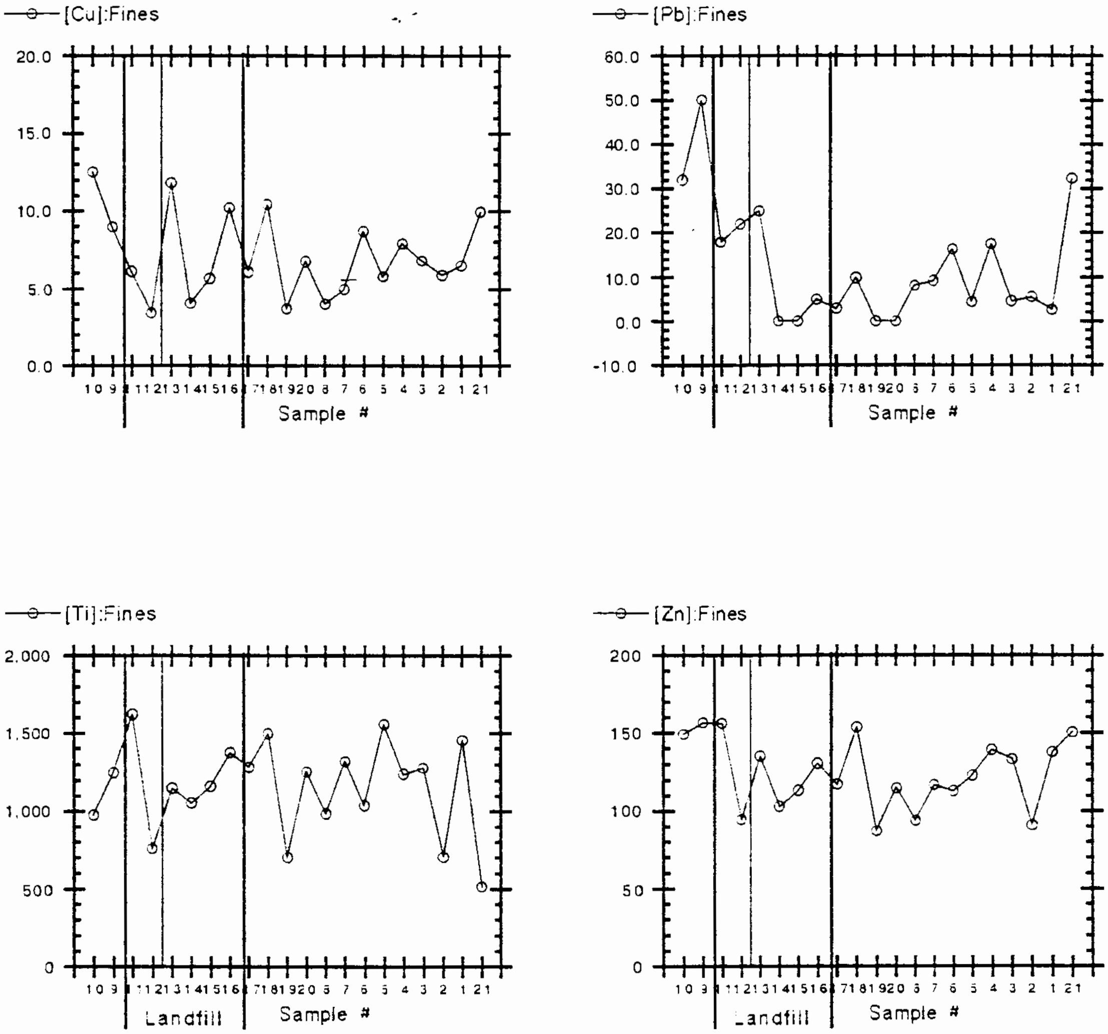


Figure 14: Frazier Brook Grab Samples (June, 1990).
Trace Metals to Percent Fines Normalized Values.

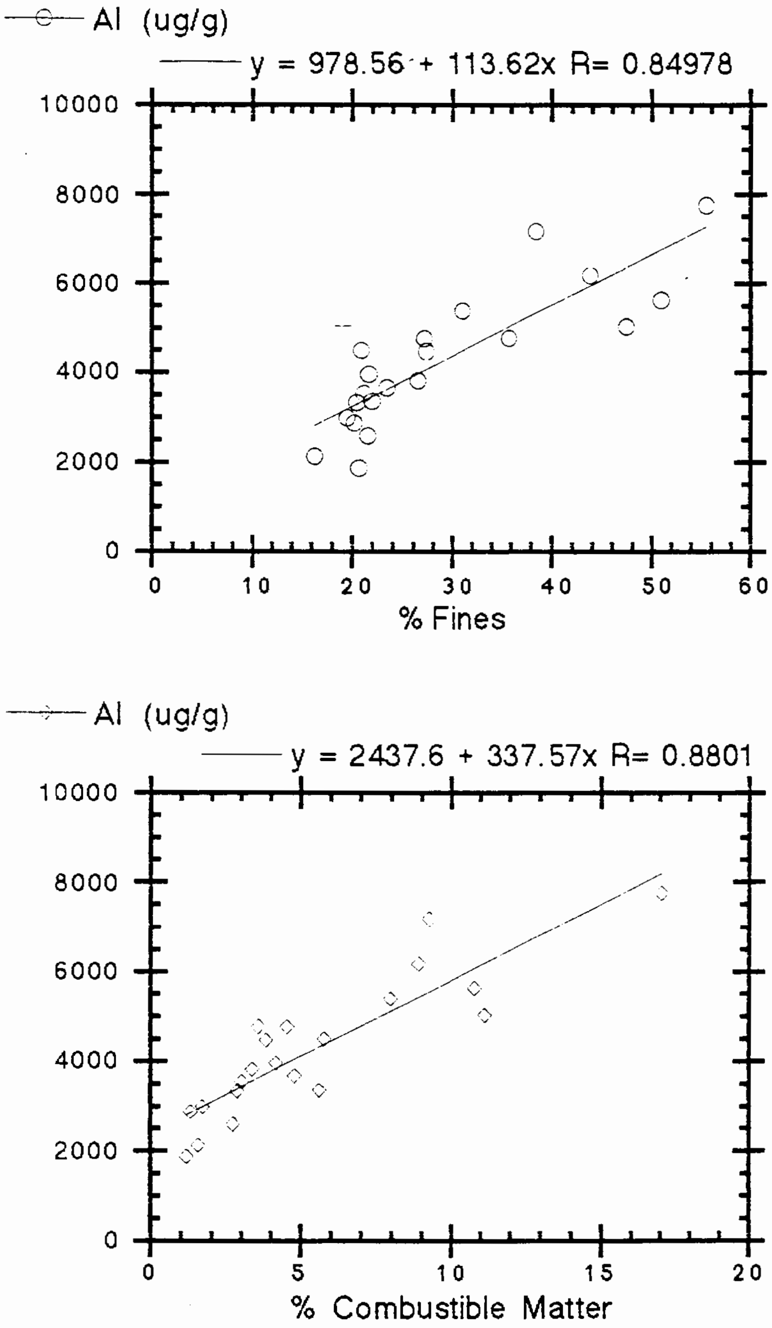


Figure 15. Linear Regression Example.
Al vs %Fines and Al vs %Combustible Matter
in Frazier Brook Sediment Samples (June, 1990).

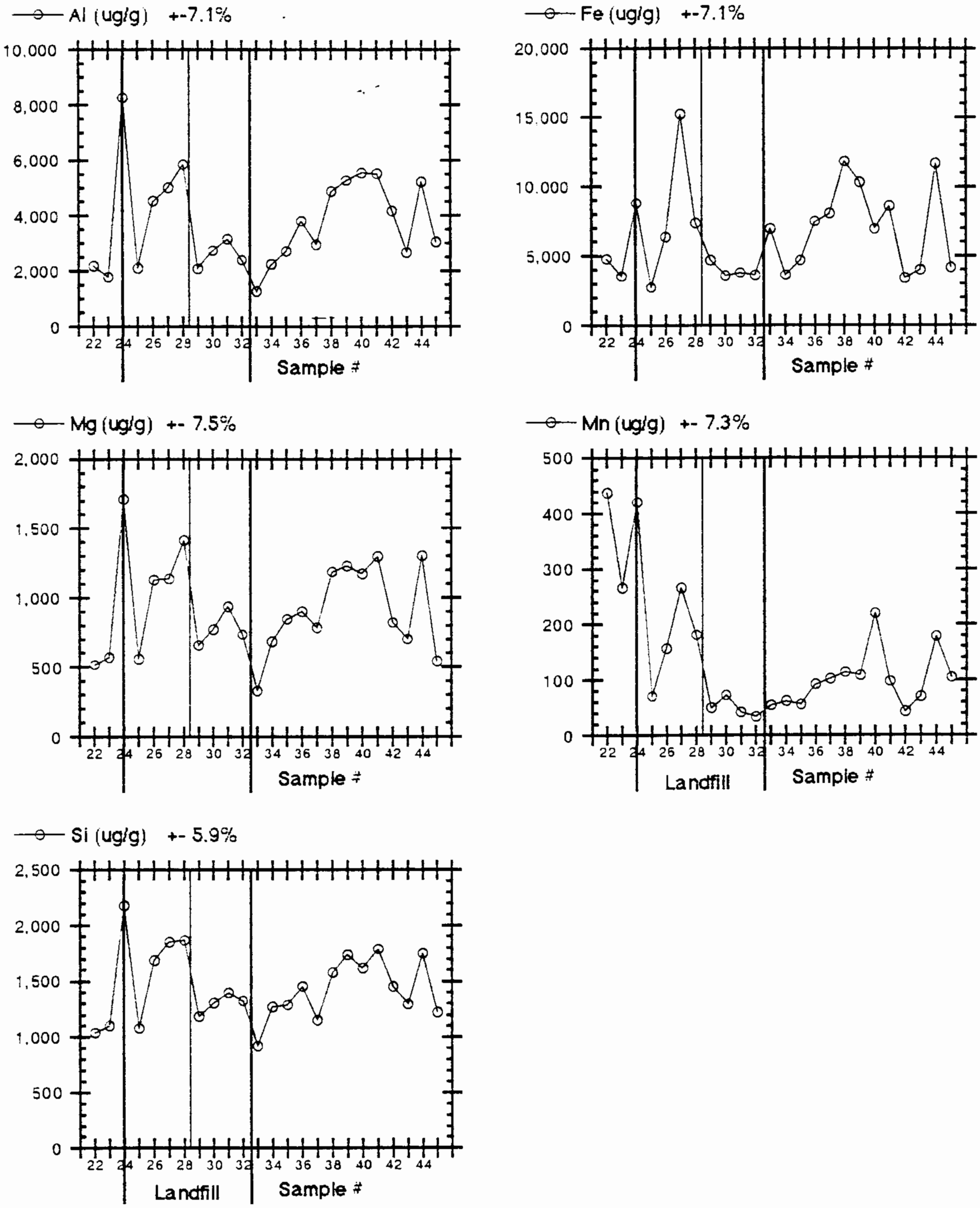


Figure 16: Frazier Brook Grab Samples (October,1990). Major Metal Concentration Per Gram Sediment. Circled points represent additional sampling locations in Frazier Brook above the Walker Brook Confluence.

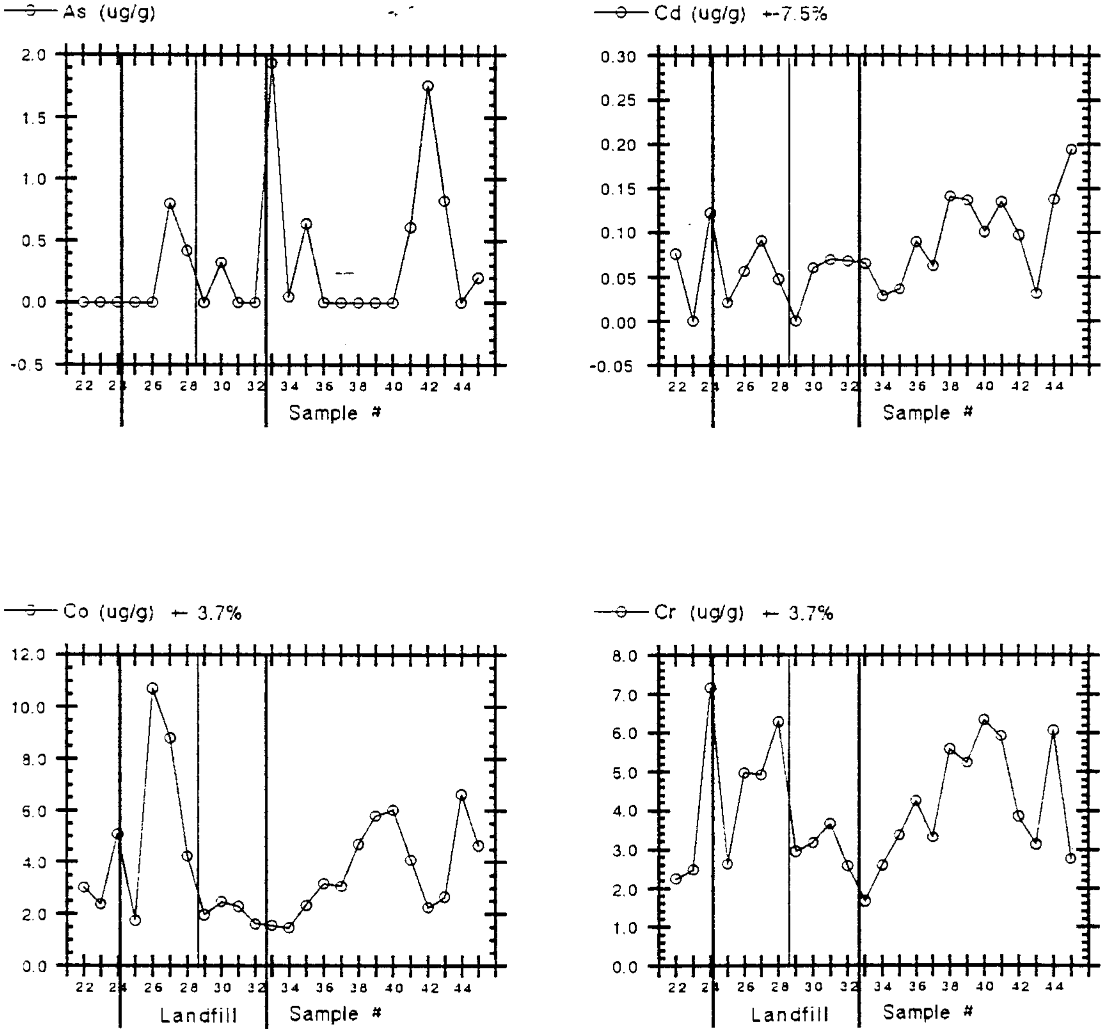


Figure 16: Frazier Brook Grab Samples (October,1990). Trace Metal Concentration Per Gram Sediment. Circled points represent additional sampling locations in Frazier Brook above the Walker Brook Confluence.

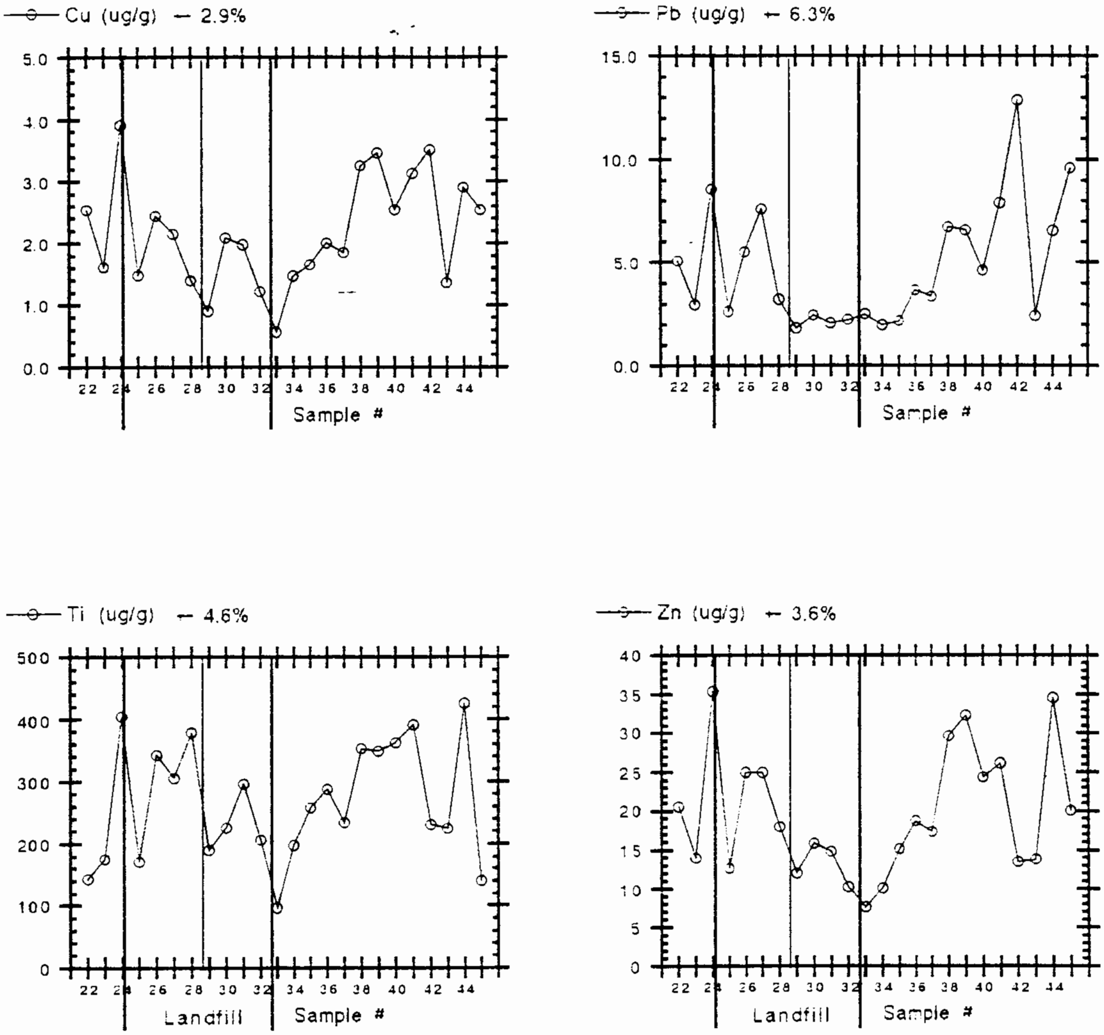


Figure 16: Frazier Brook Grab Samples (October,1990).
Trace Metal Concentration Per Gram Sediment.
Circled points represent additional sampling locations in Frazier Brook above the Walker Brook Confluence.

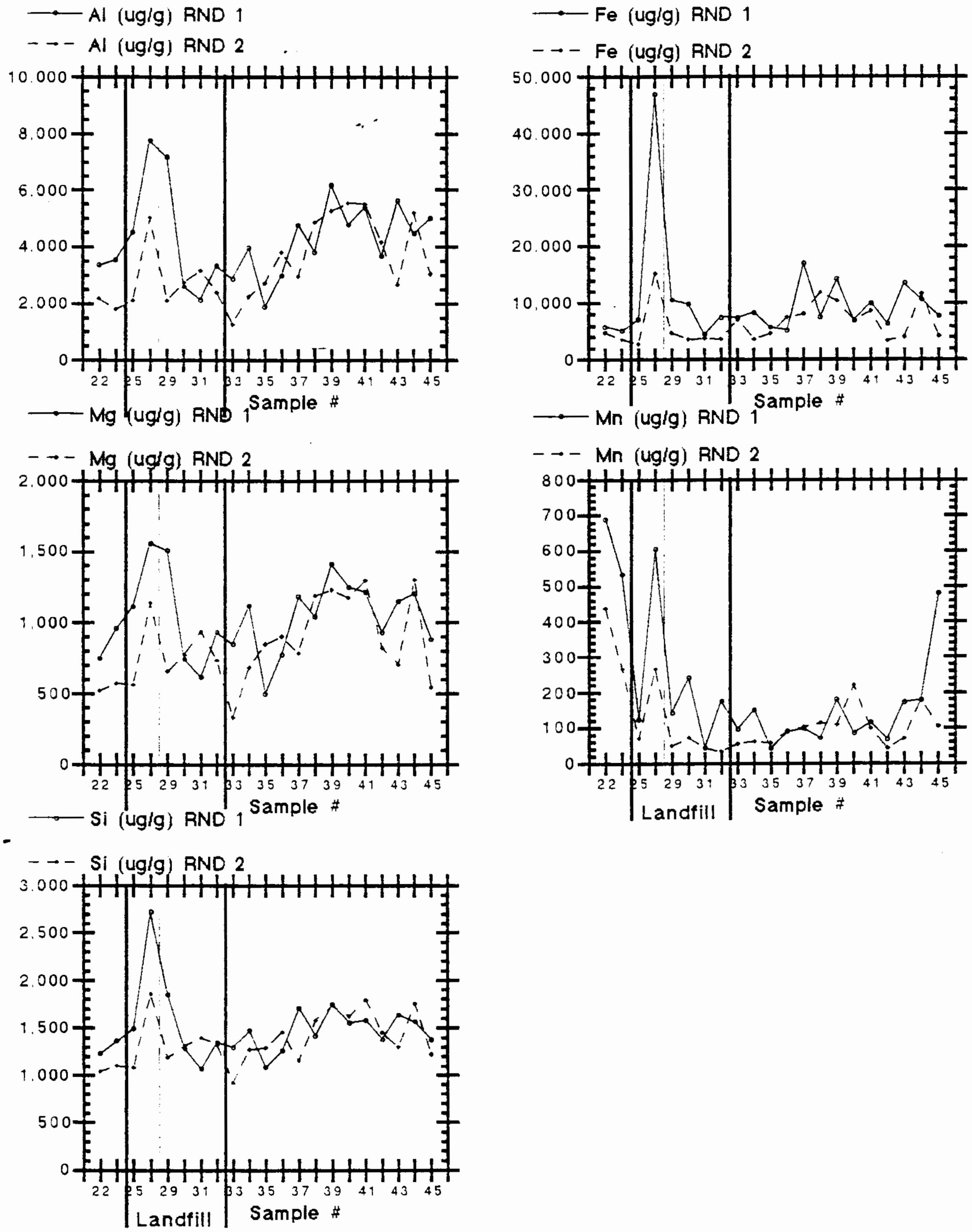


Figure 17: Frazier Brook Grab Samples
(June vs October,1990).
Major Metal Concentration Per Gram Sediment.

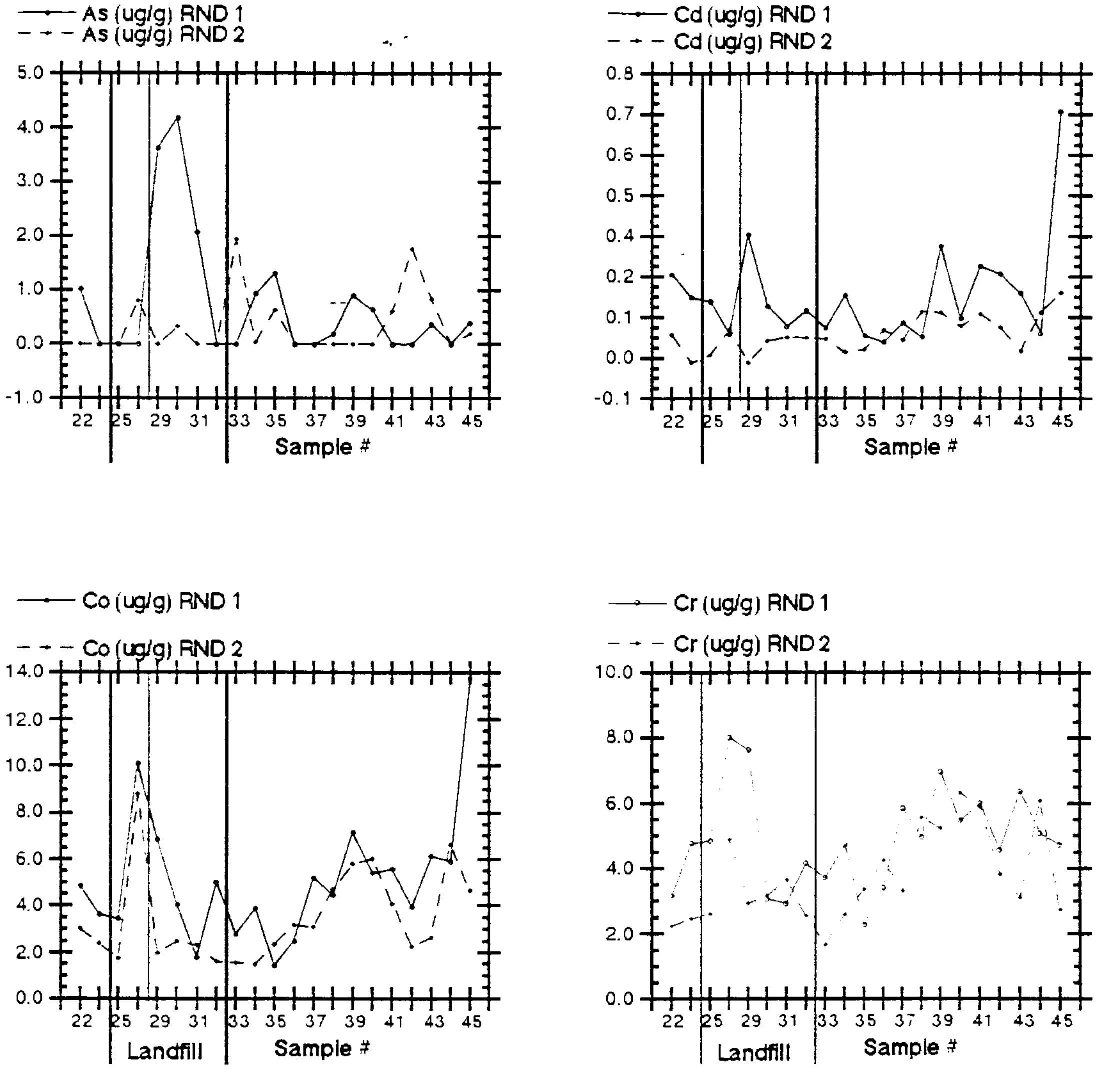


Figure 17: Frazier Brook Grab Samples
(June vs October, 1990).
Trace Metal Concentration Per Gram Sediment.

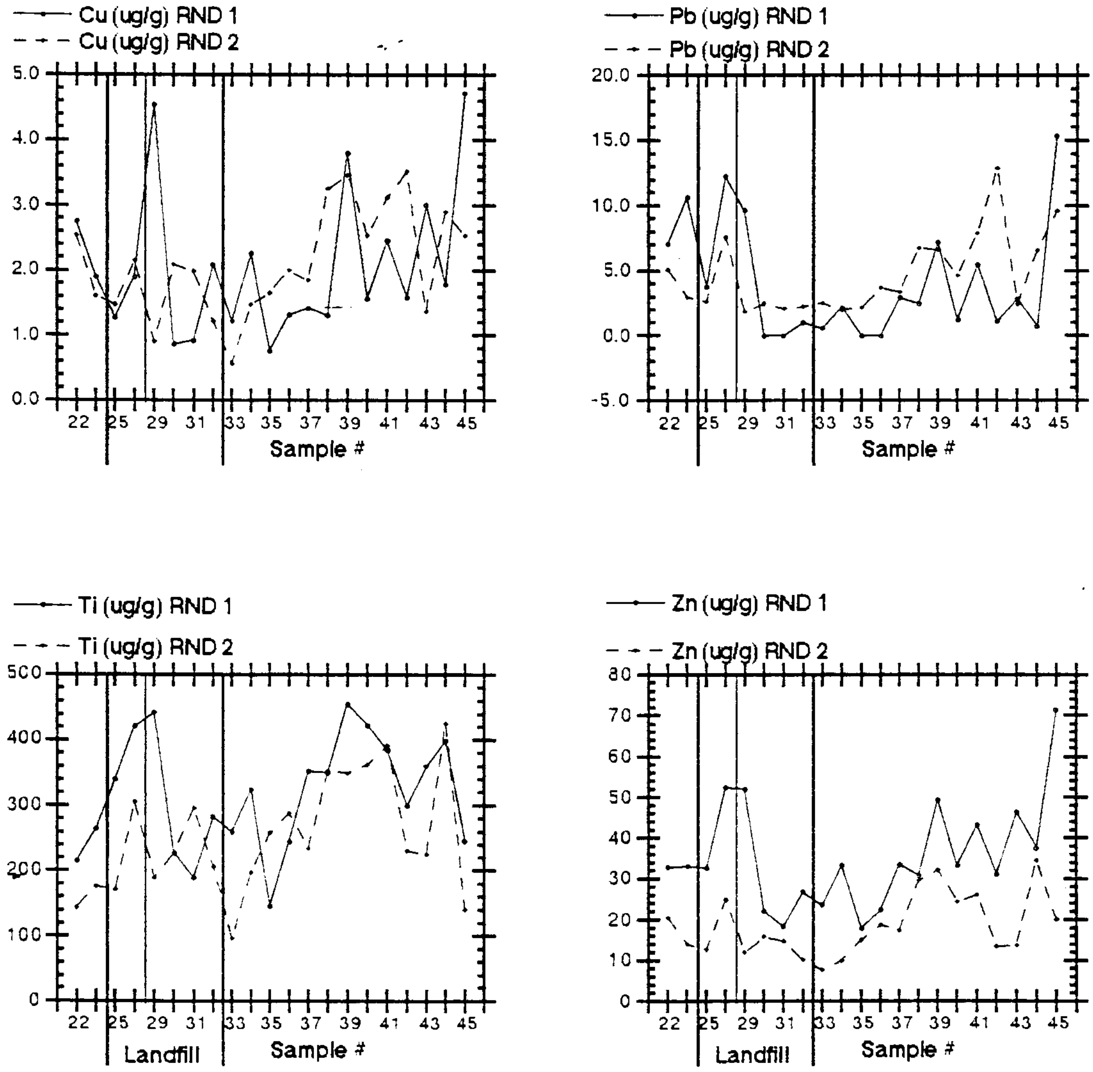


Figure 17: Frazier Brook Grab Samples
(June vs October, 1990).
Trace Metal Concentration Per Gram Sediment.

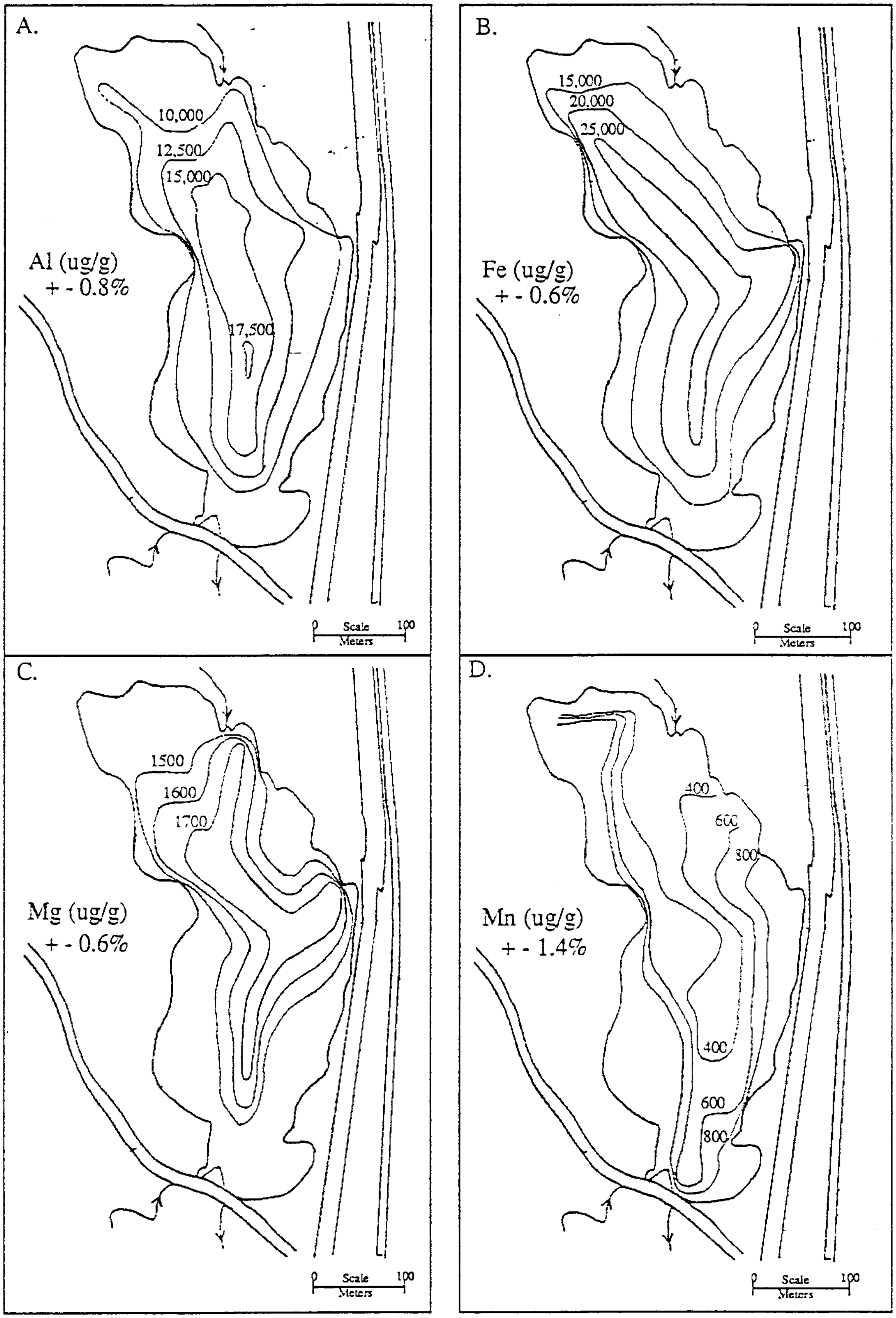


Fig 18: Metal Concentration Contours in Eagle Pond Sediments; June, 1990.

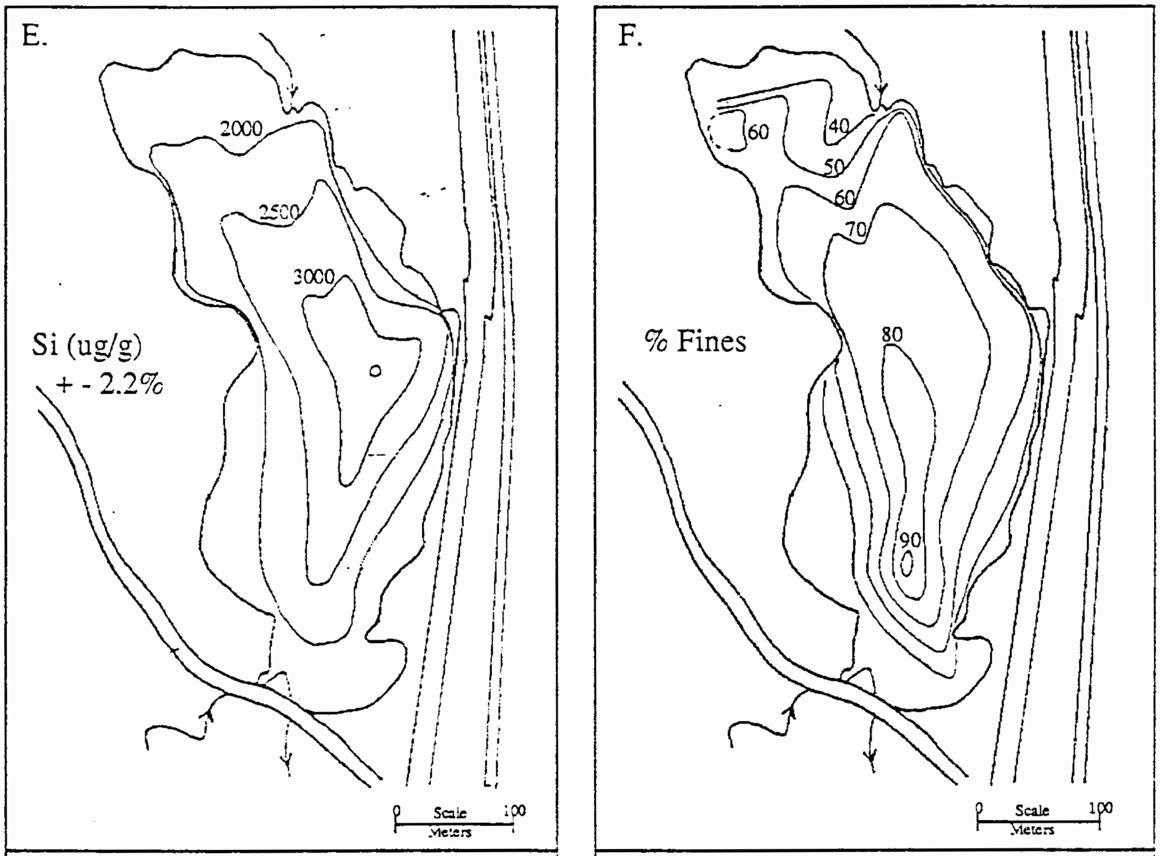


Fig 18: Metal Concentration Contours in Eagle Pond Sediments; June, 1990.

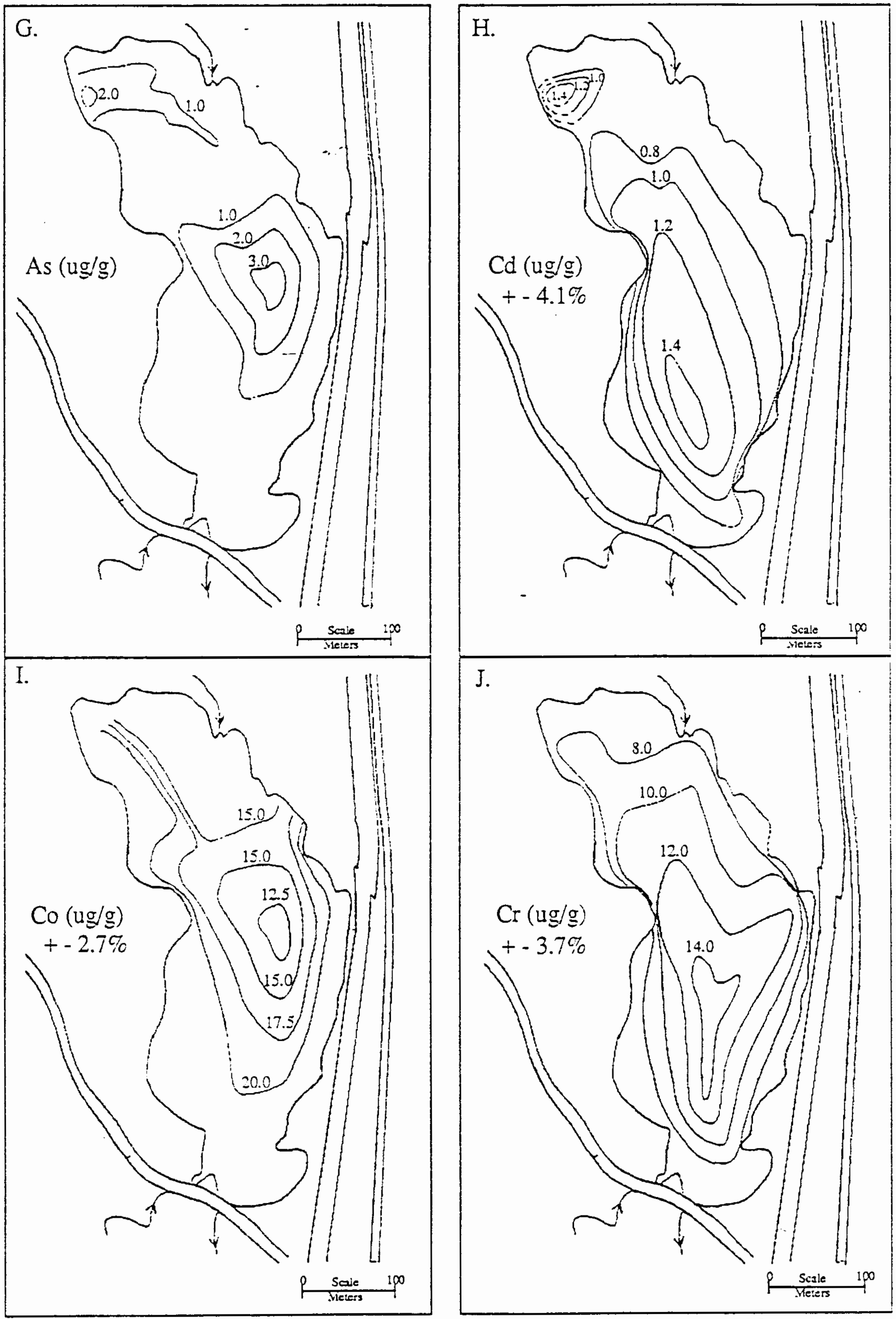


Fig 18: Metal Concentration Contours in Eagle Pond Sediments; June, 1990.

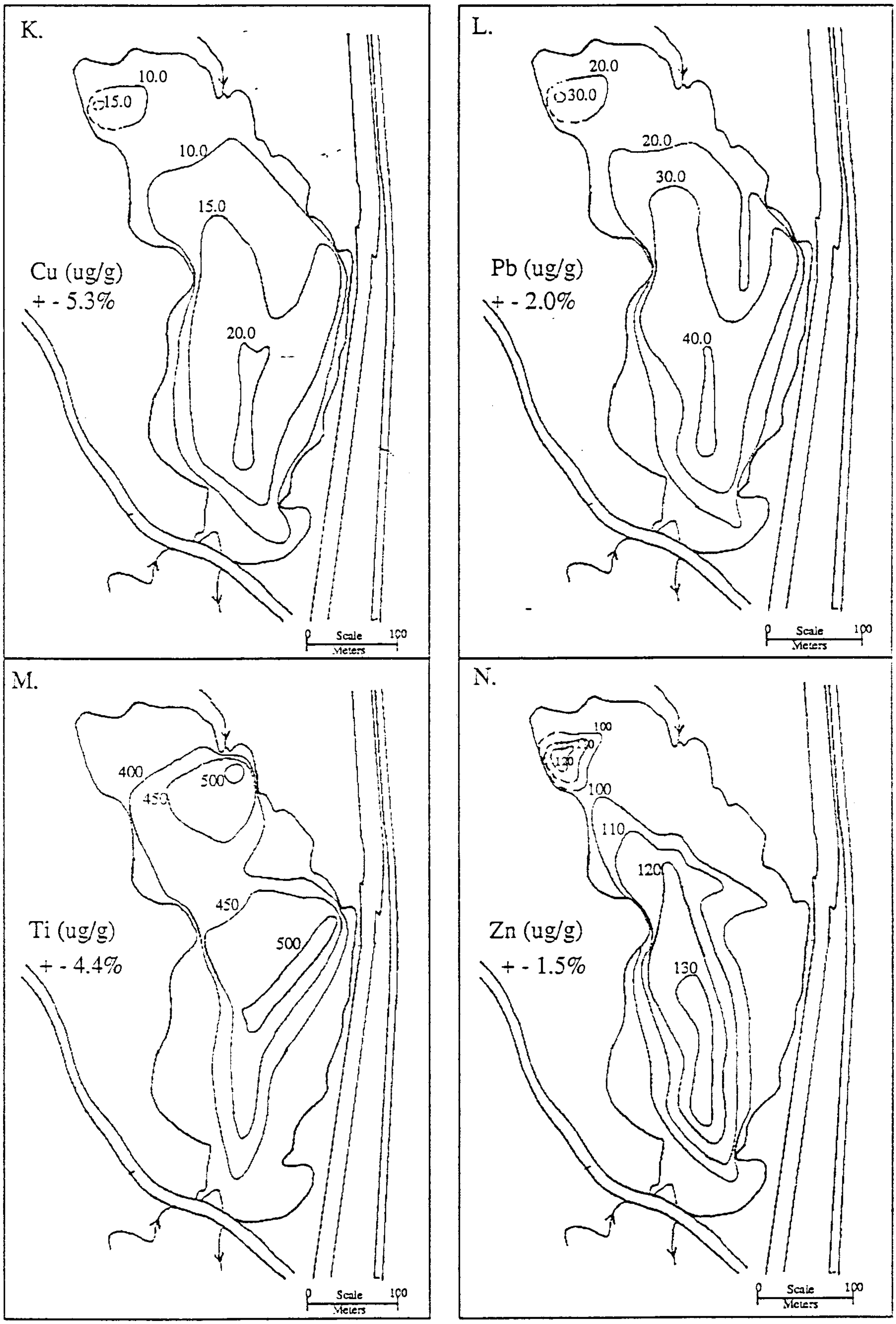


Fig 18: Metal Concentration Contours in Eagle Pond Sediments; June, 1990.

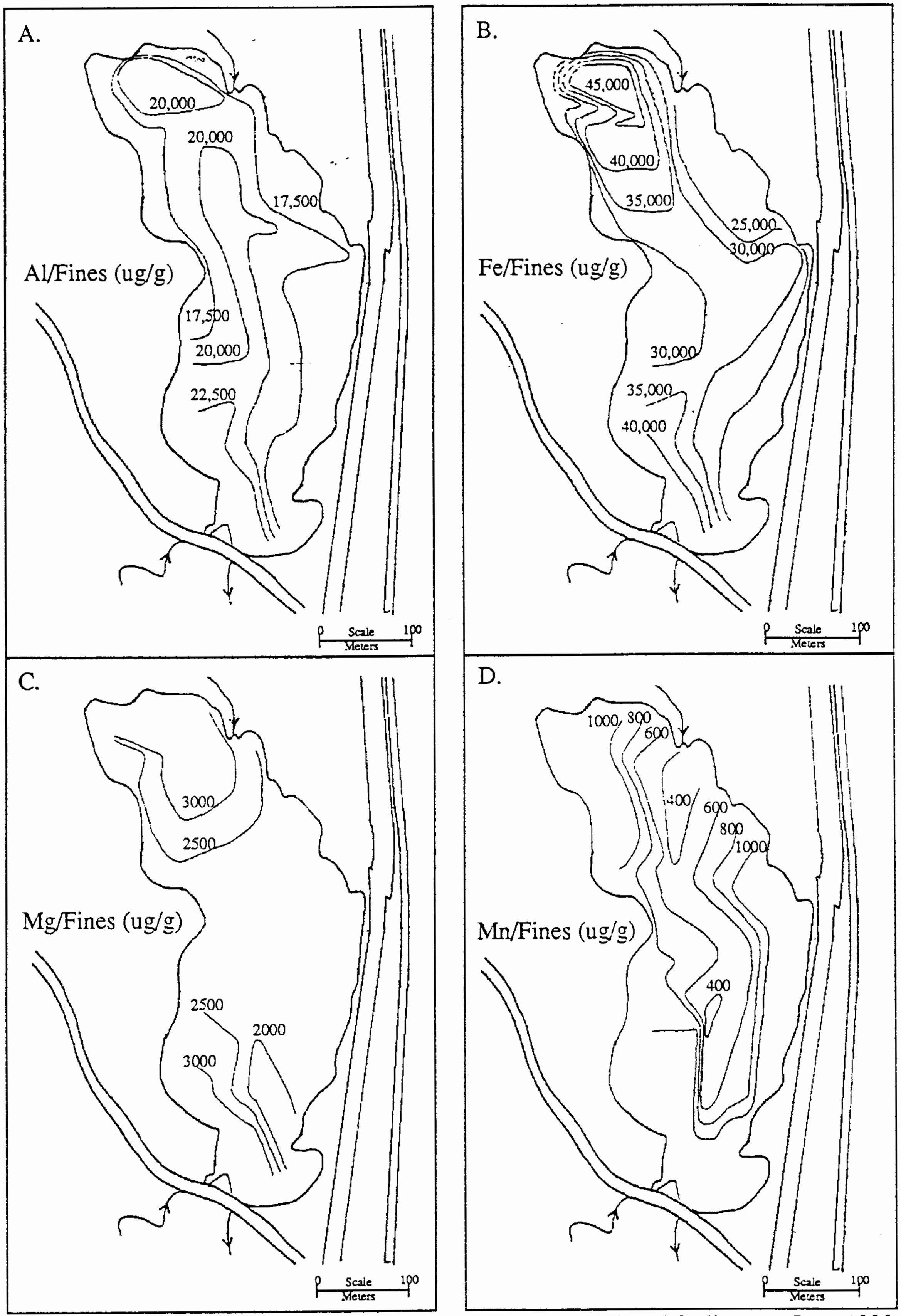


Fig 19: Metal Concentration to Percent Fines Ratios in Eagle Pond Sediments; June,1990.

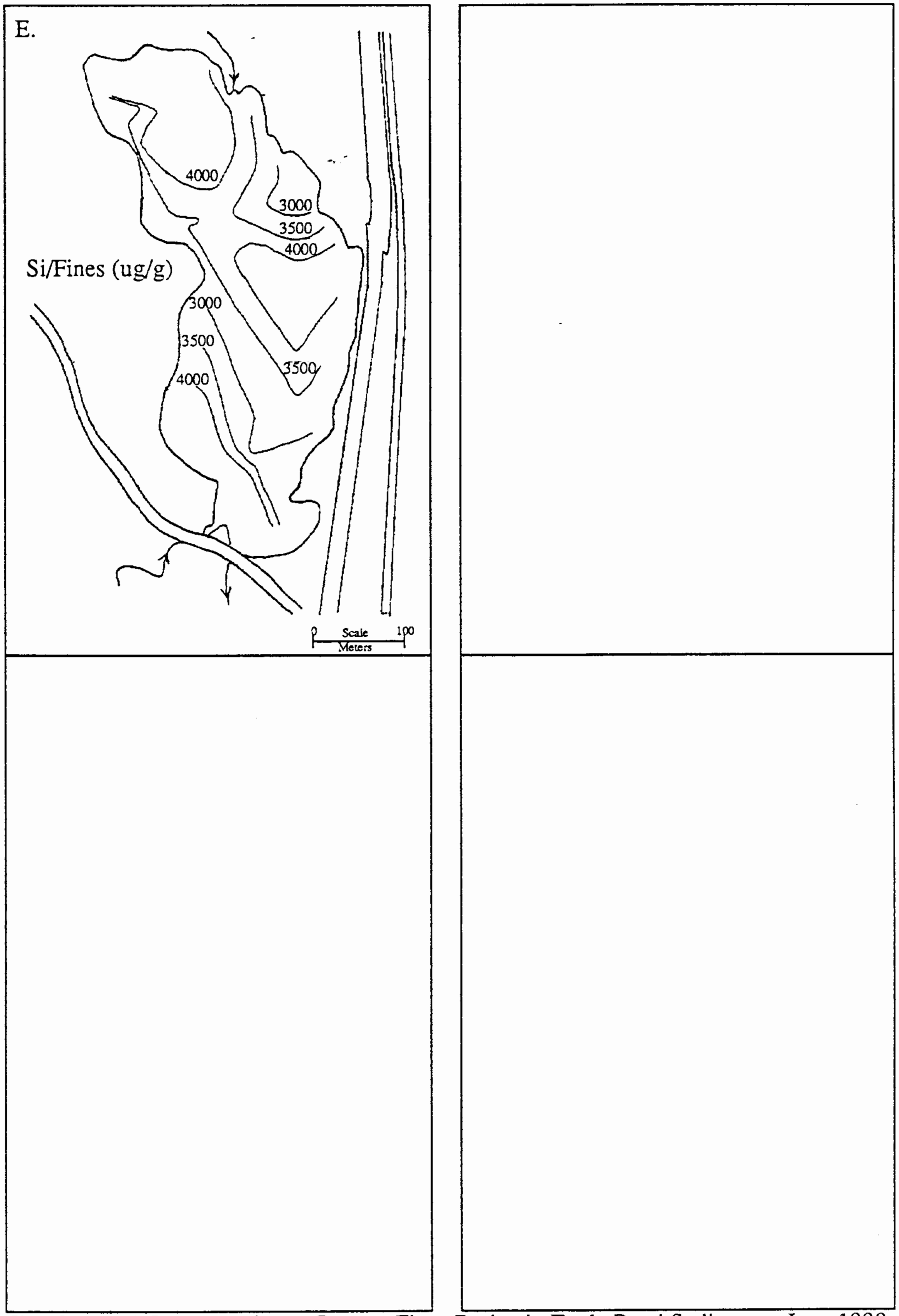


Fig 19: Metal Concentration to Percent Fines Ratios in Eagle Pond Sediments; June,1990.

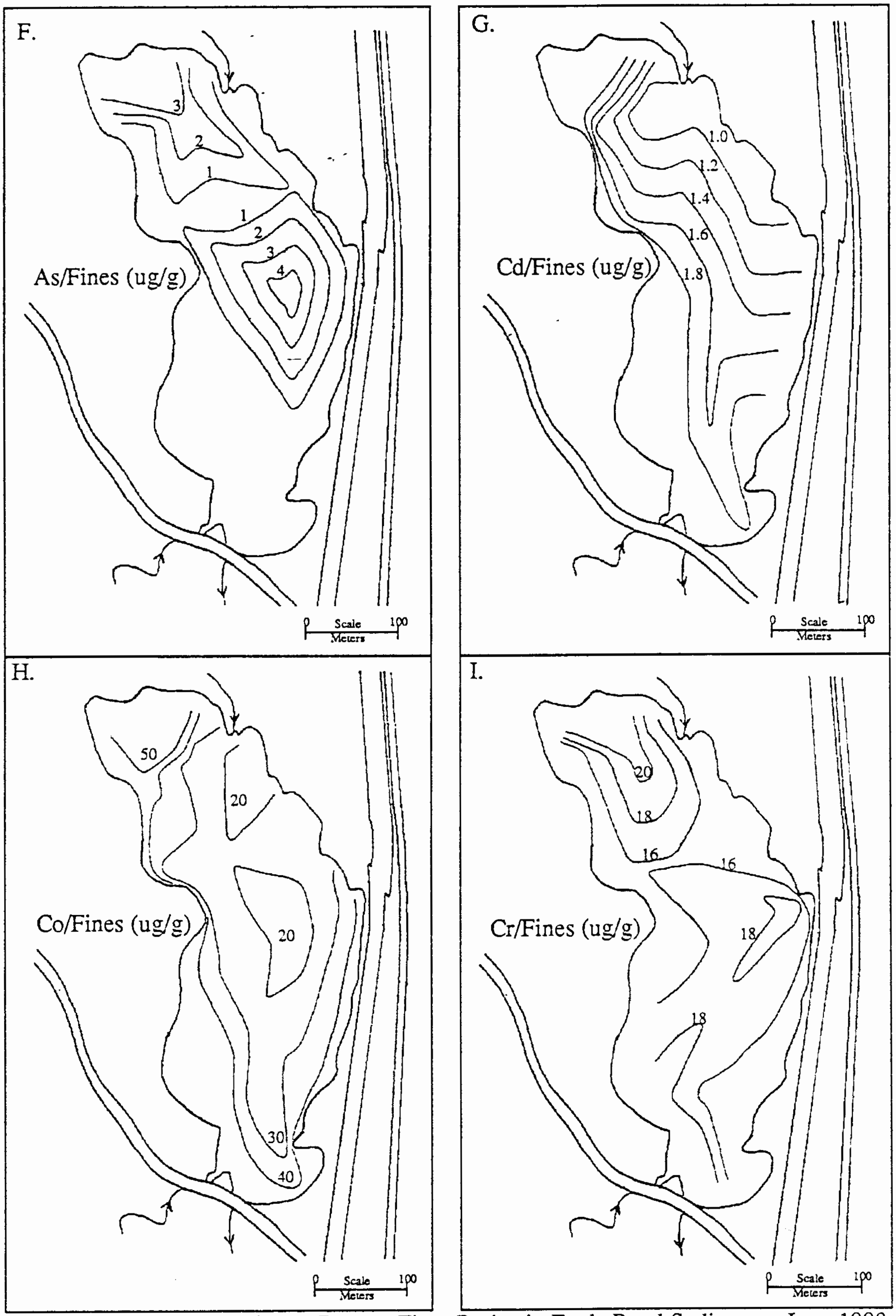


Fig 19: Metal Concentration to Percent Fines Ratios in Eagle Pond Sediments; June,1990.

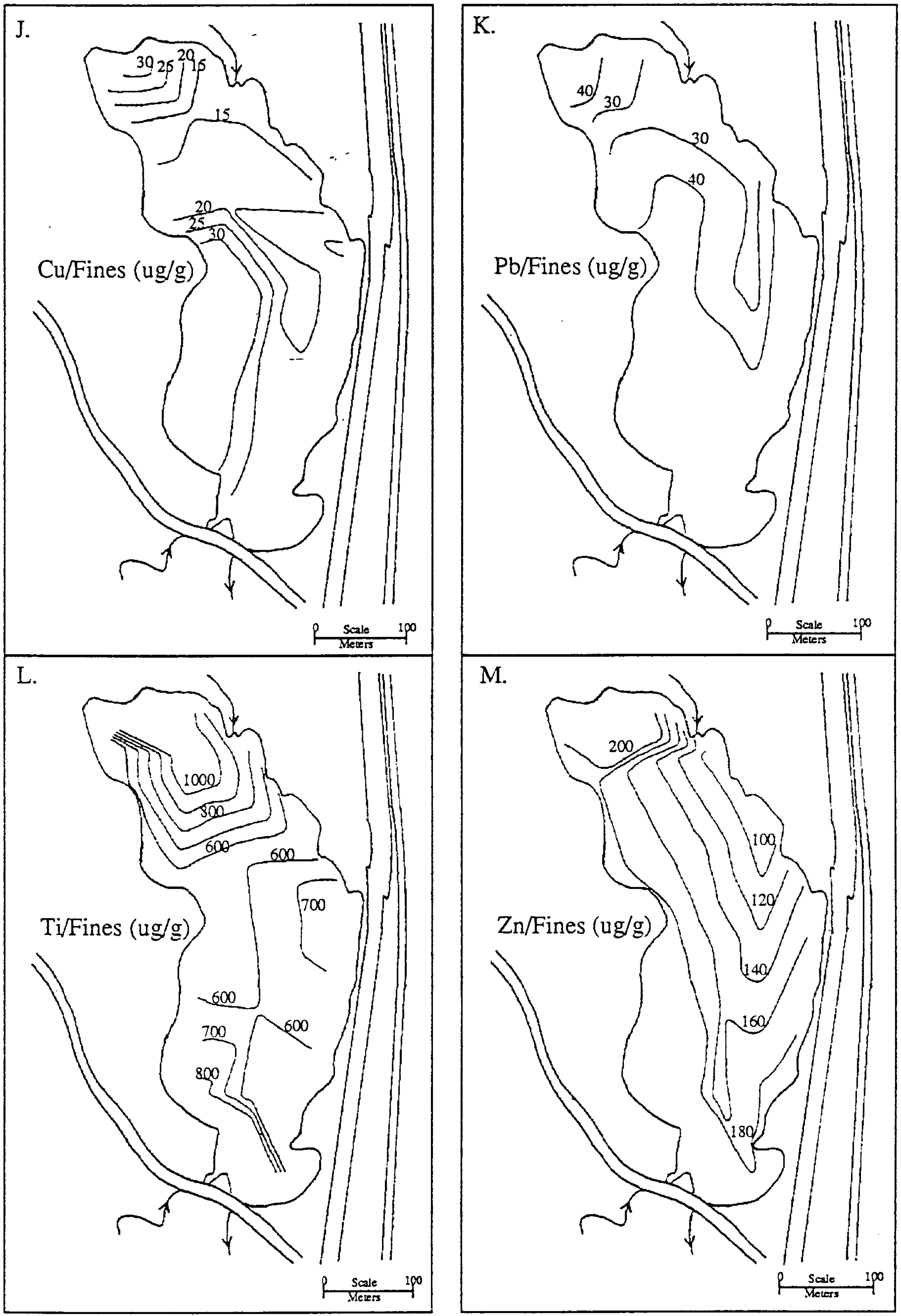


Fig 19: Metal Concentration to Percent Fines Ratios in Eagle Pond Sediments; June,1990.

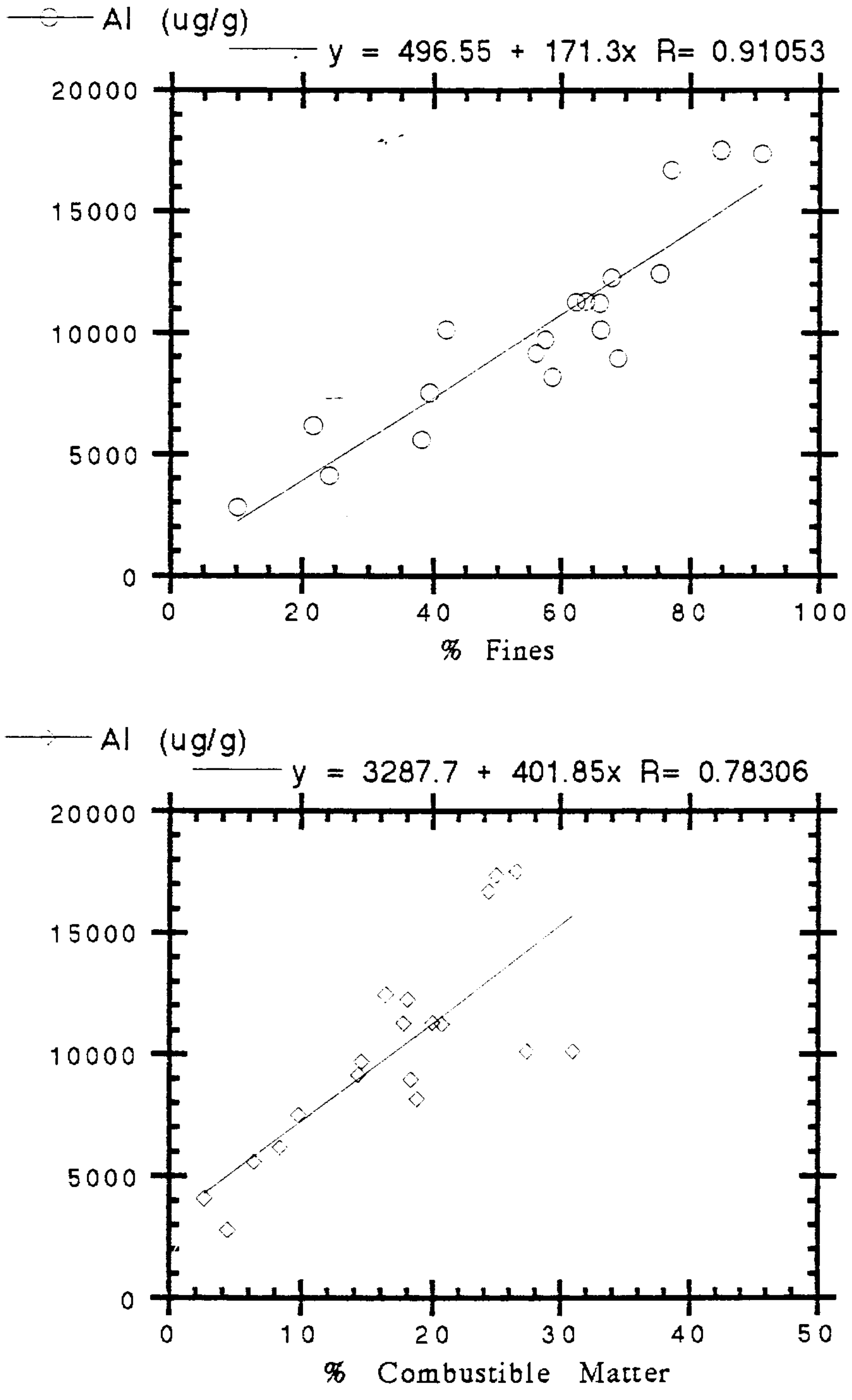


Figure 20: Linear Regression Example.
Al vs %Fines and Al vs %Combustible Matter
in Eagle Pond Bottom Samples (June, 1990).

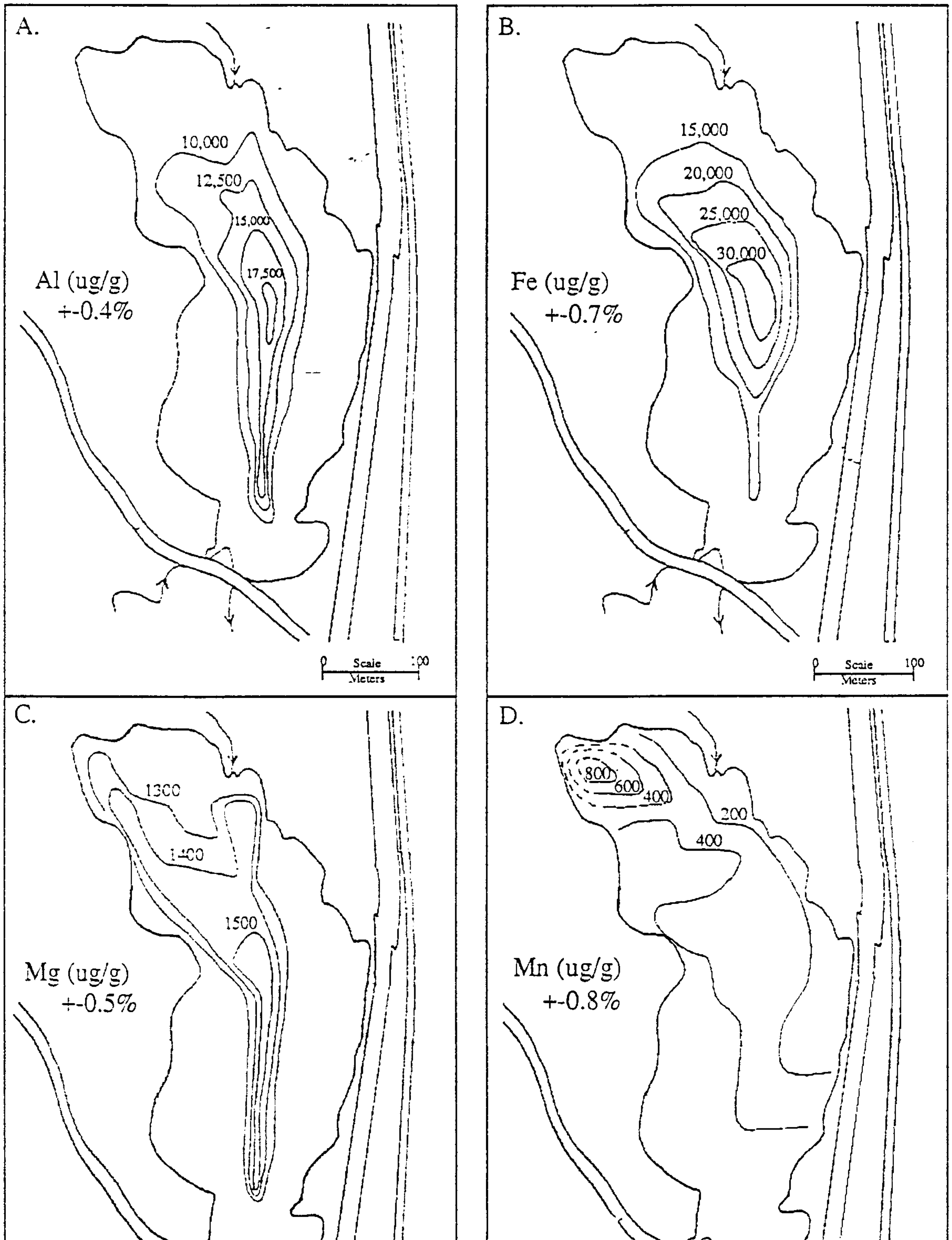


Fig 21: Metal Concentration Contours in Eagle Pond Sediments; October, 1990.

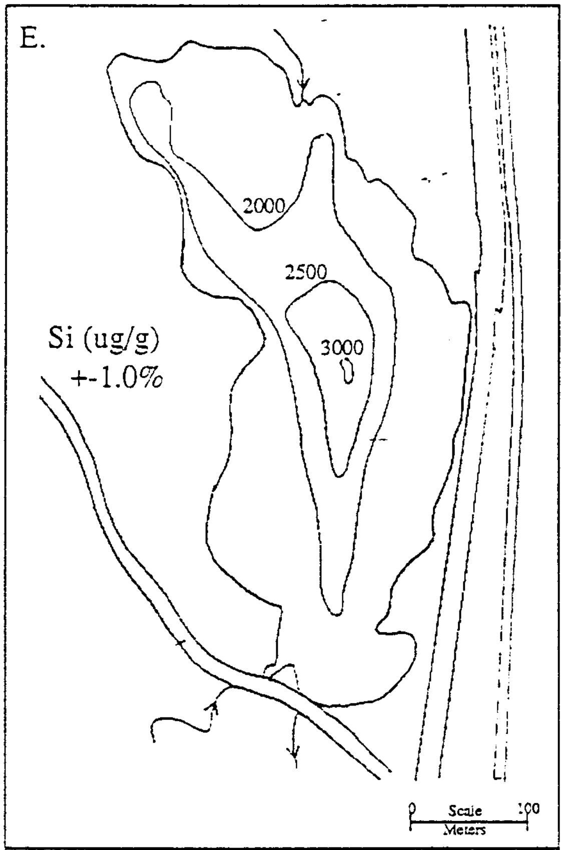


Fig 21: Metal Concentration Contours in Eagle Pond Sediments; October, 1990.

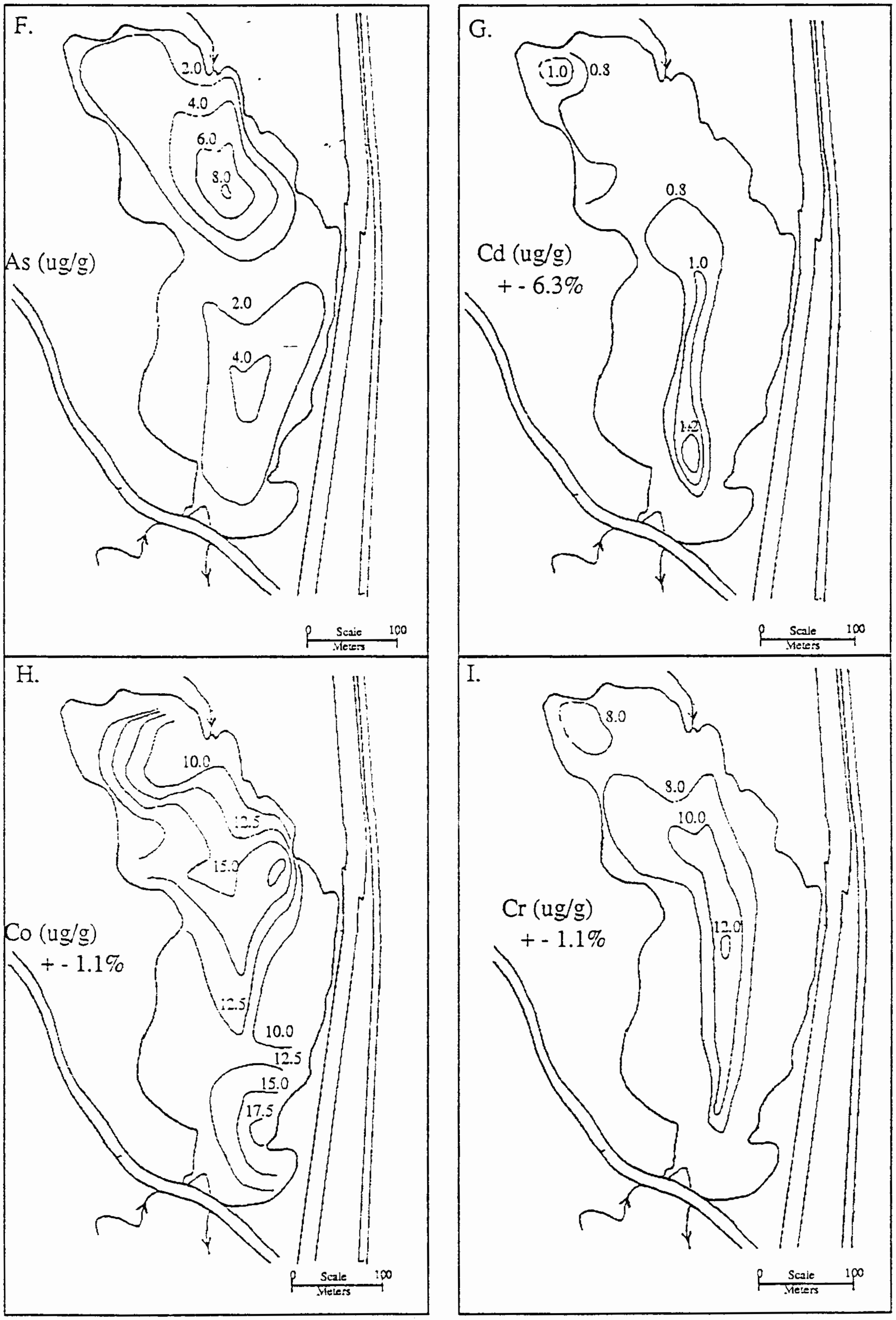


Fig 21: Metal Concentration Contours in Eagle Pond Sediments; October, 1990.

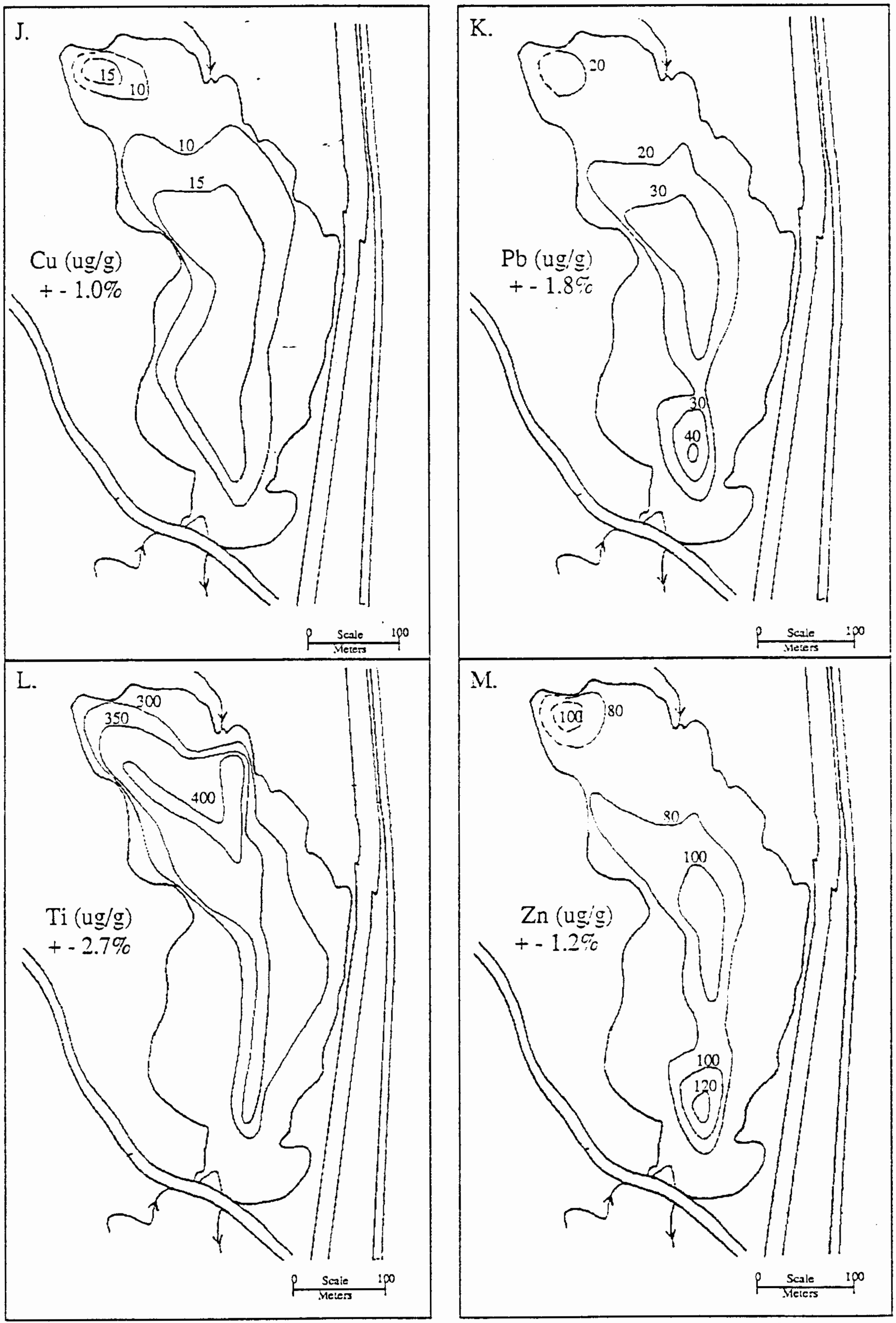


Fig 21: Metal Concentration Contours in Eagle Pond Sediments; October, 1990

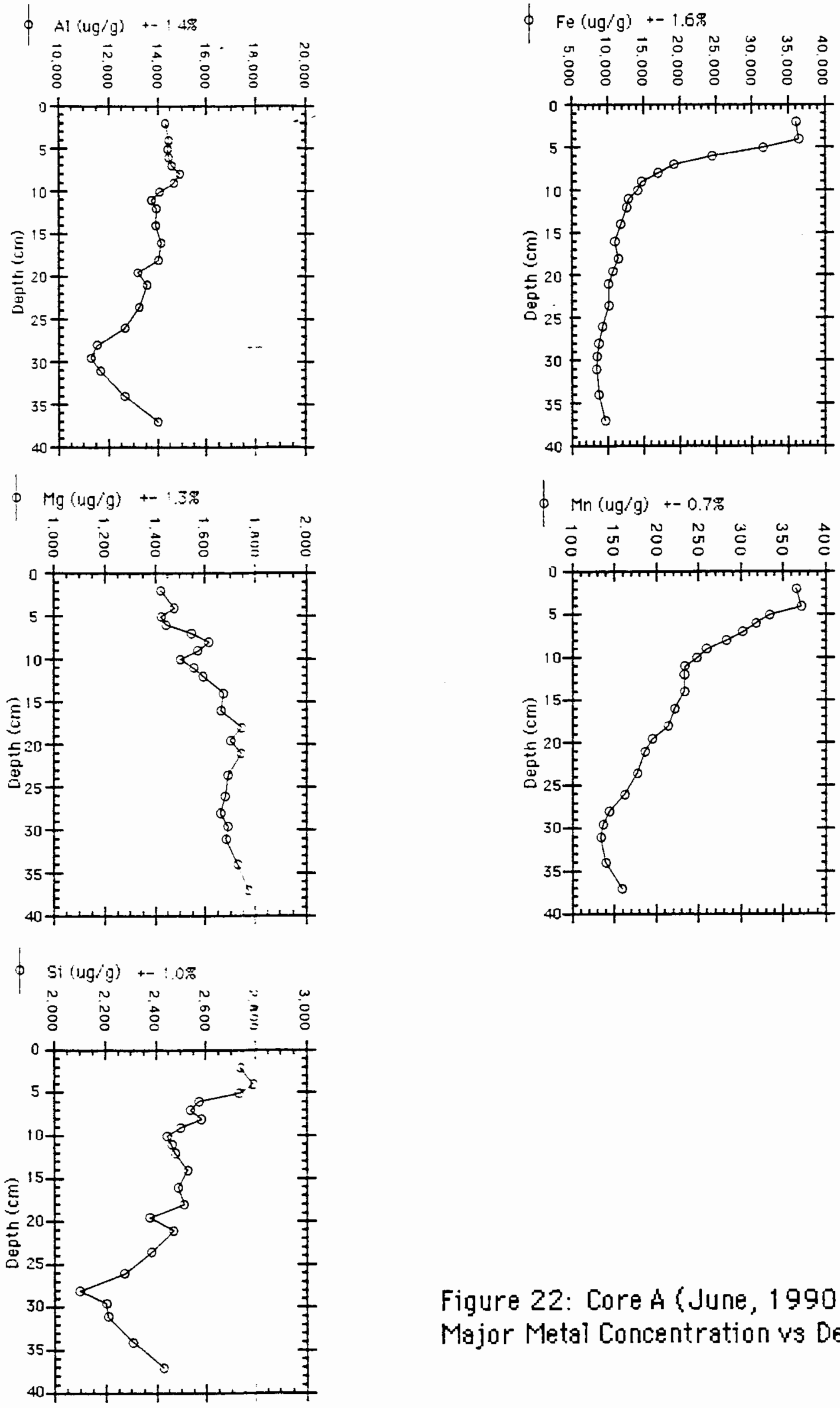


Figure 22: Core A (June, 1990). Major Metal Concentration vs Depth.

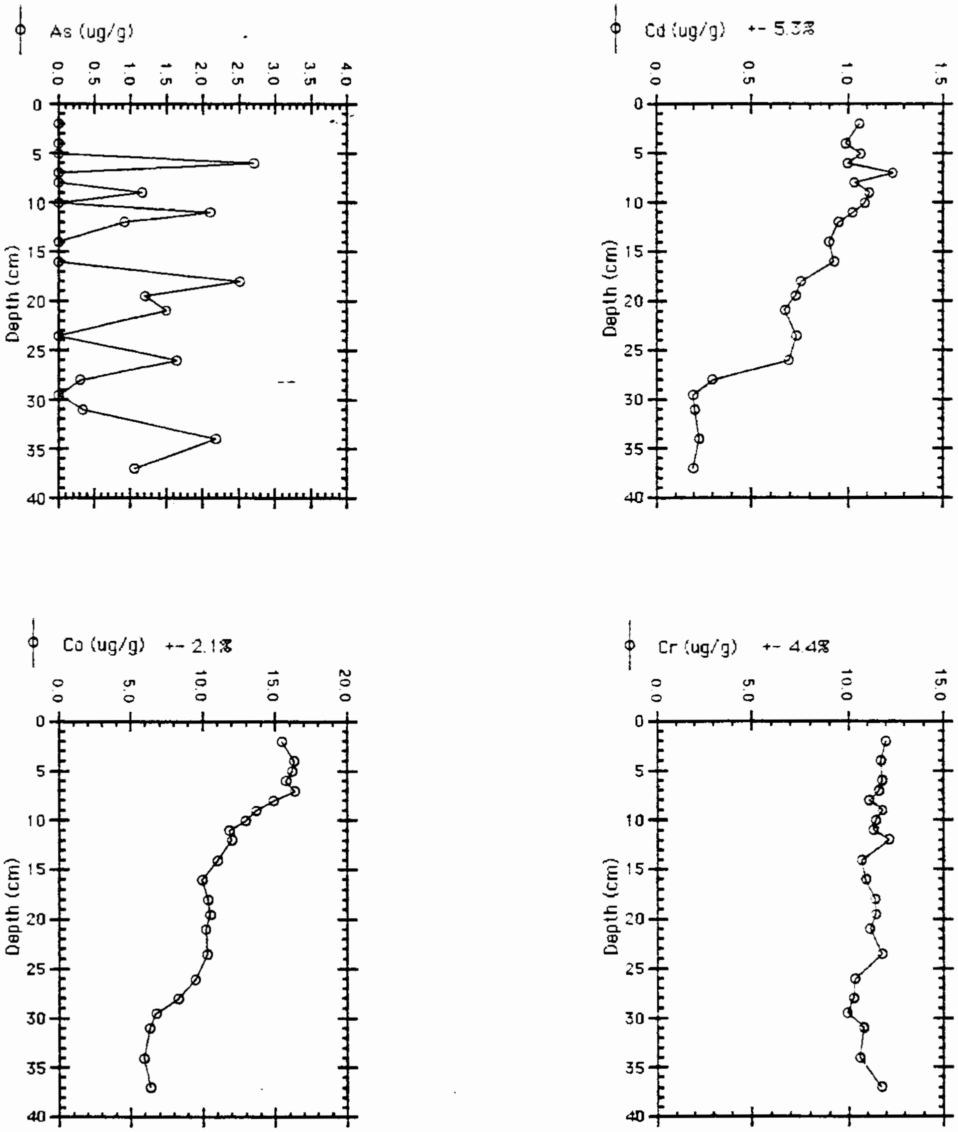


Figure 22: Core A (June, 1990).
Trace Metal Concentration vs Depth.

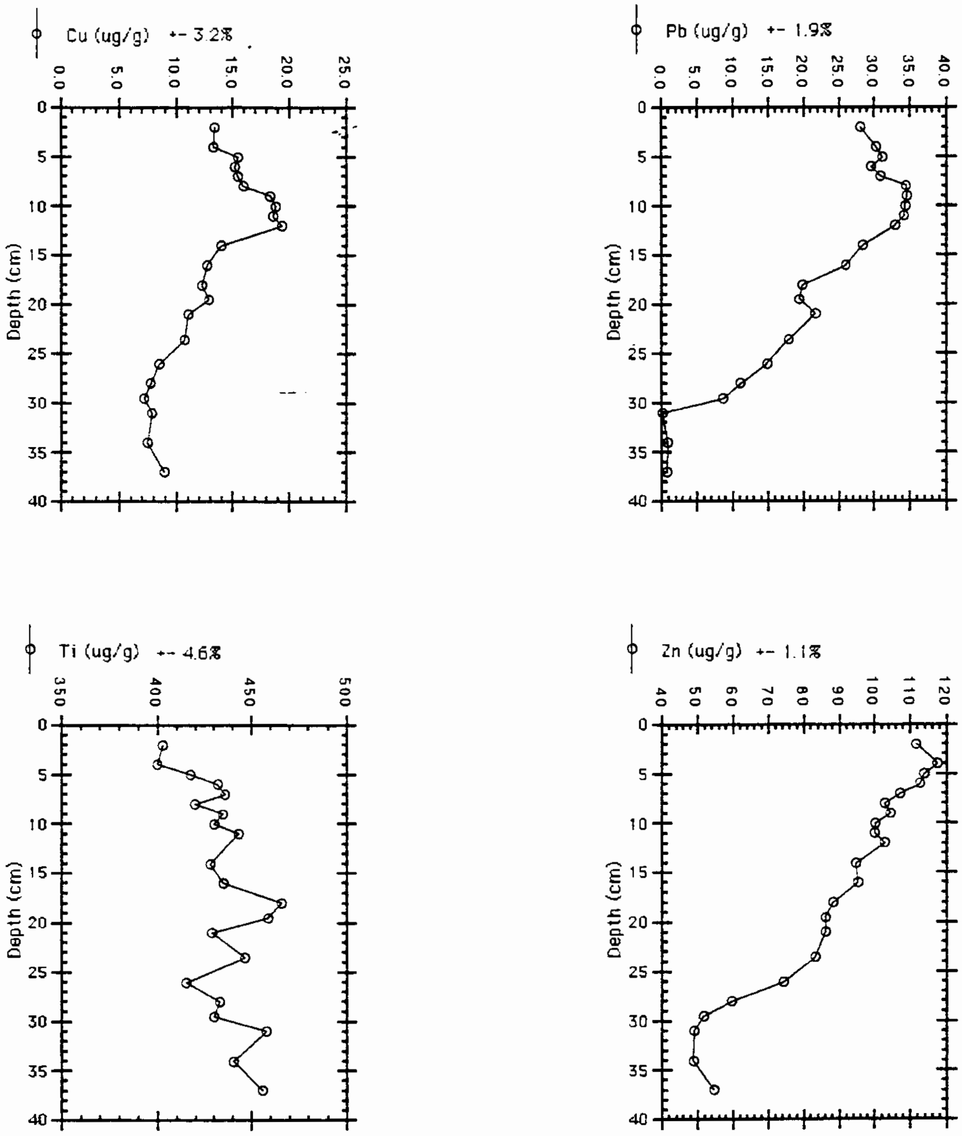


Figure 22: Core A (June, 1990).
Trace Metal Concentration vs Depth.

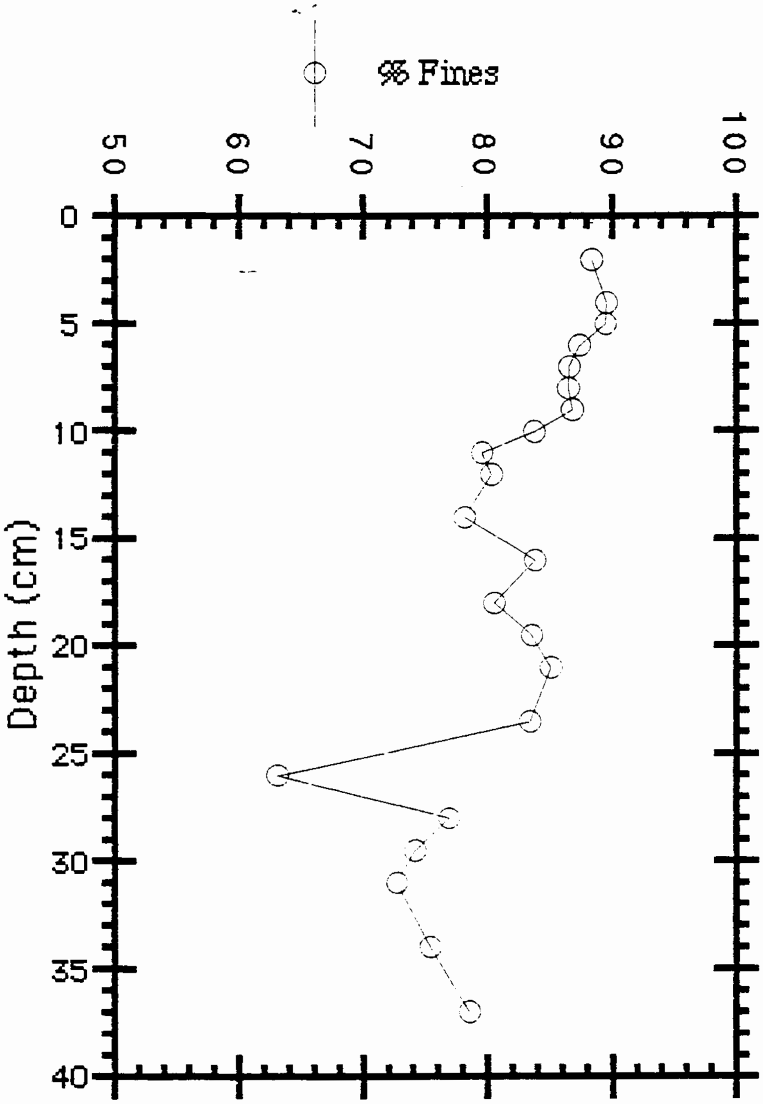


Figure 23: Core A,
Percent Fines vs Depth.

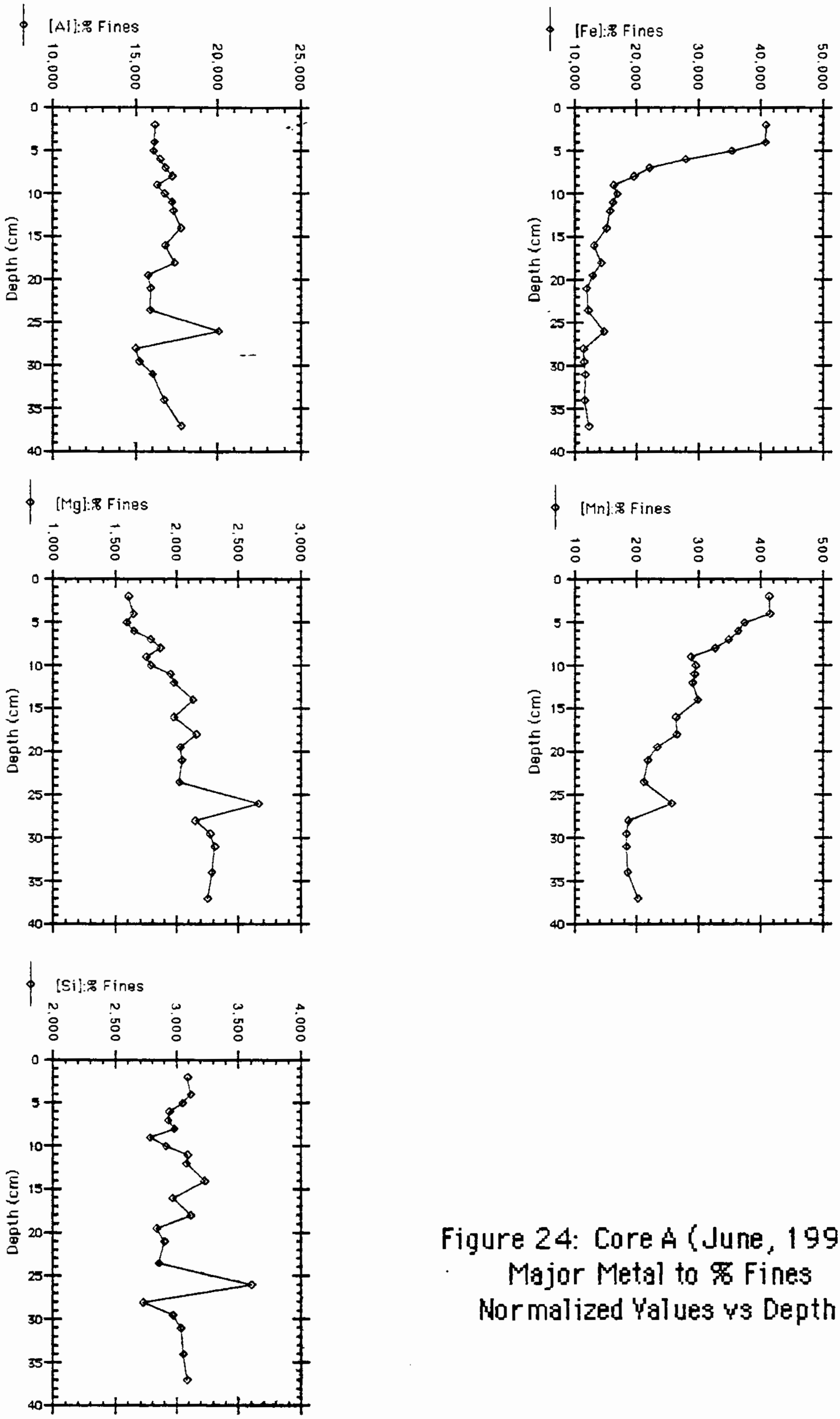


Figure 24: Core A (June, 1990).
Major Metal to % Fines
Normalized Values vs Depth.

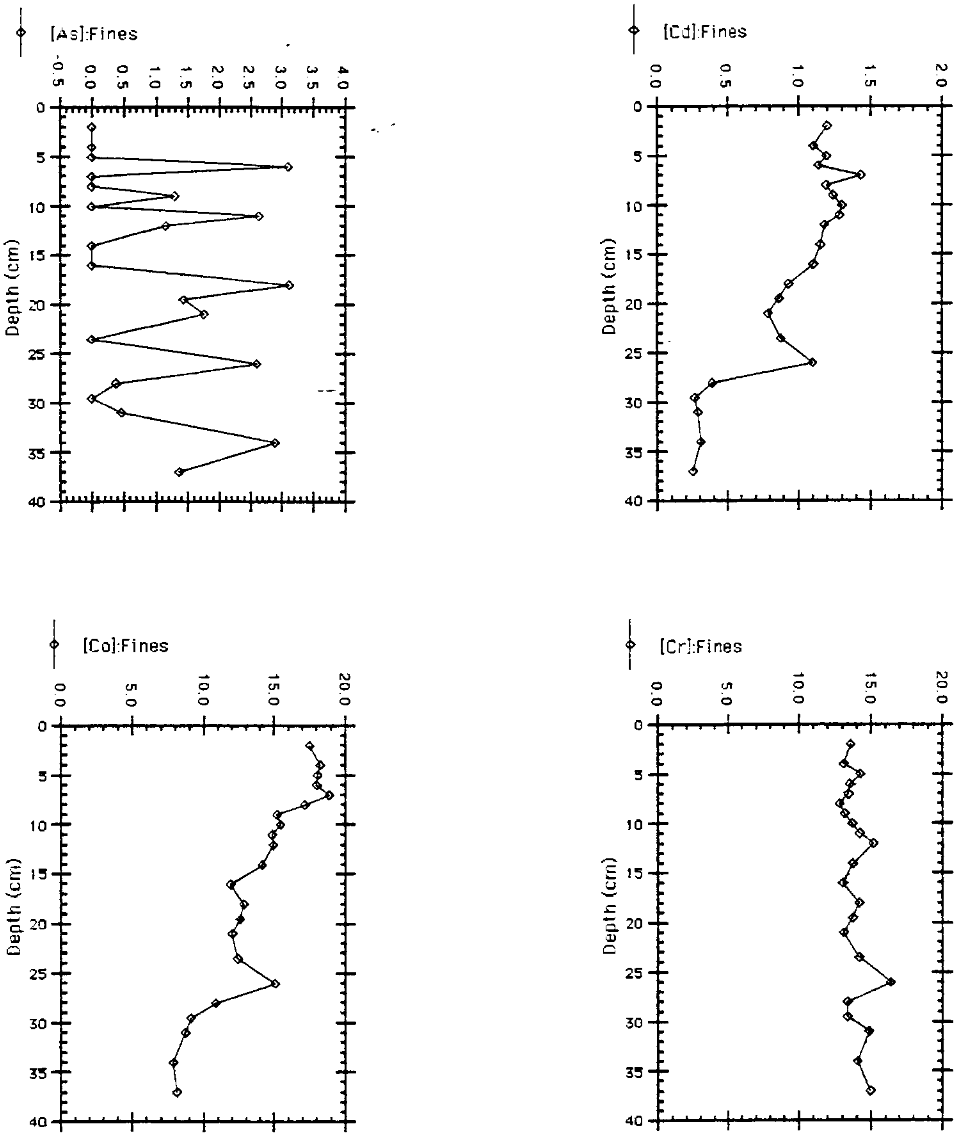


Figure 24: Core A (June, 1990).
Trace Metal to % Fines
Normalized Values vs Depth.

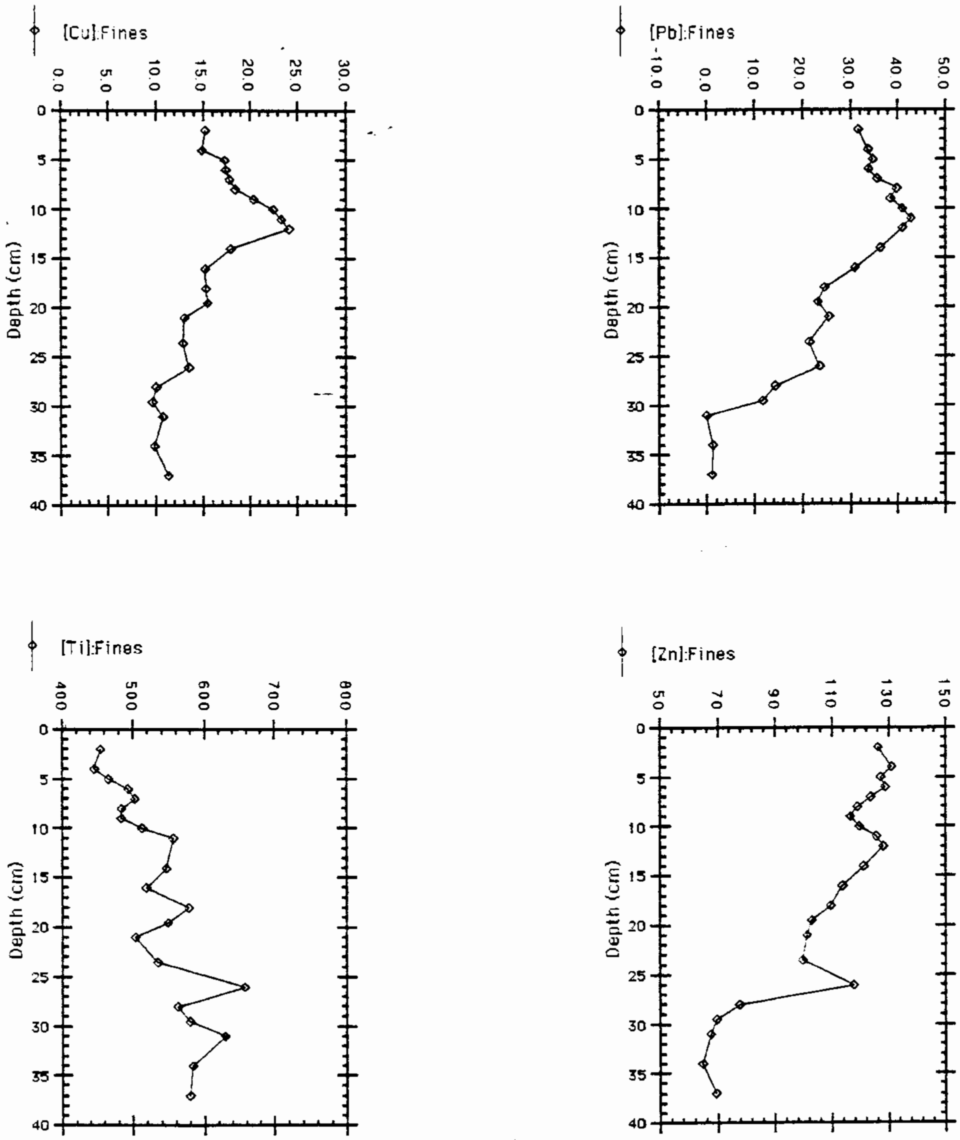


Figure 24: Core A (June, 1990).
Trace Metal to % Fines
Normalized Values vs Depth.

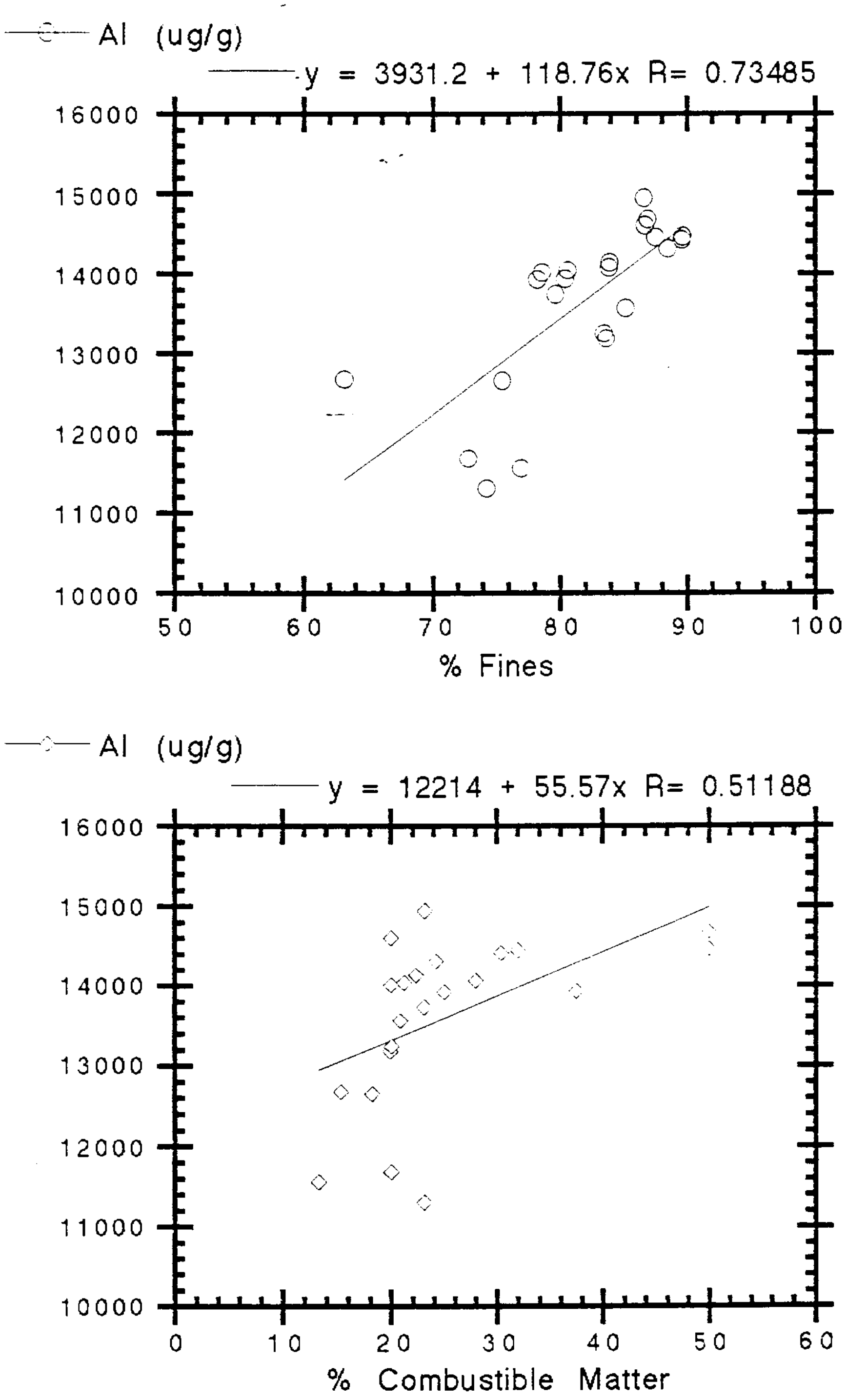


Figure 25. Linear Regression Example.
Al vs % Fines and Al vs % Combustible Matte
Core A (June, 1990).

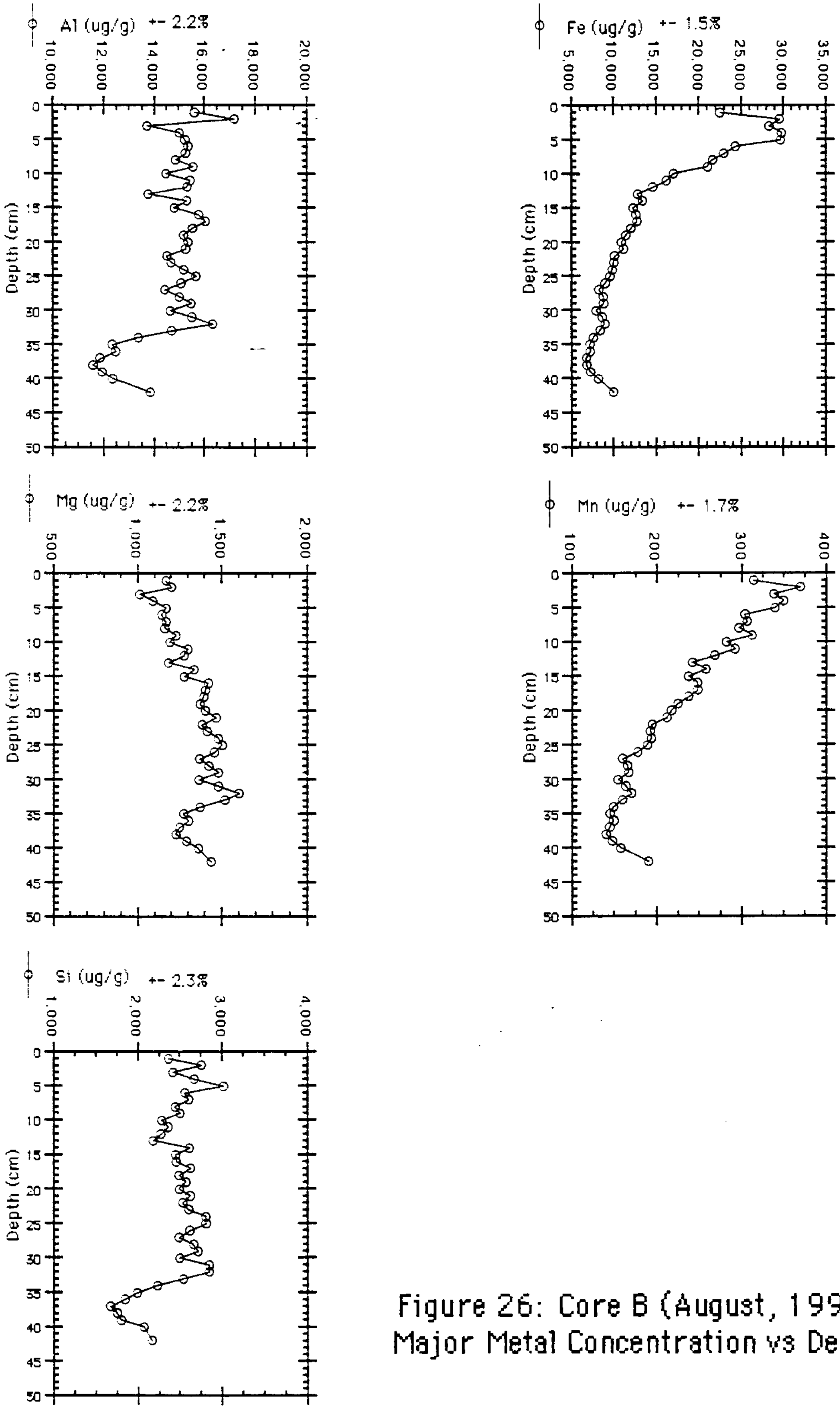


Figure 26: Core B (August, 1990). Major Metal Concentration vs Depth.

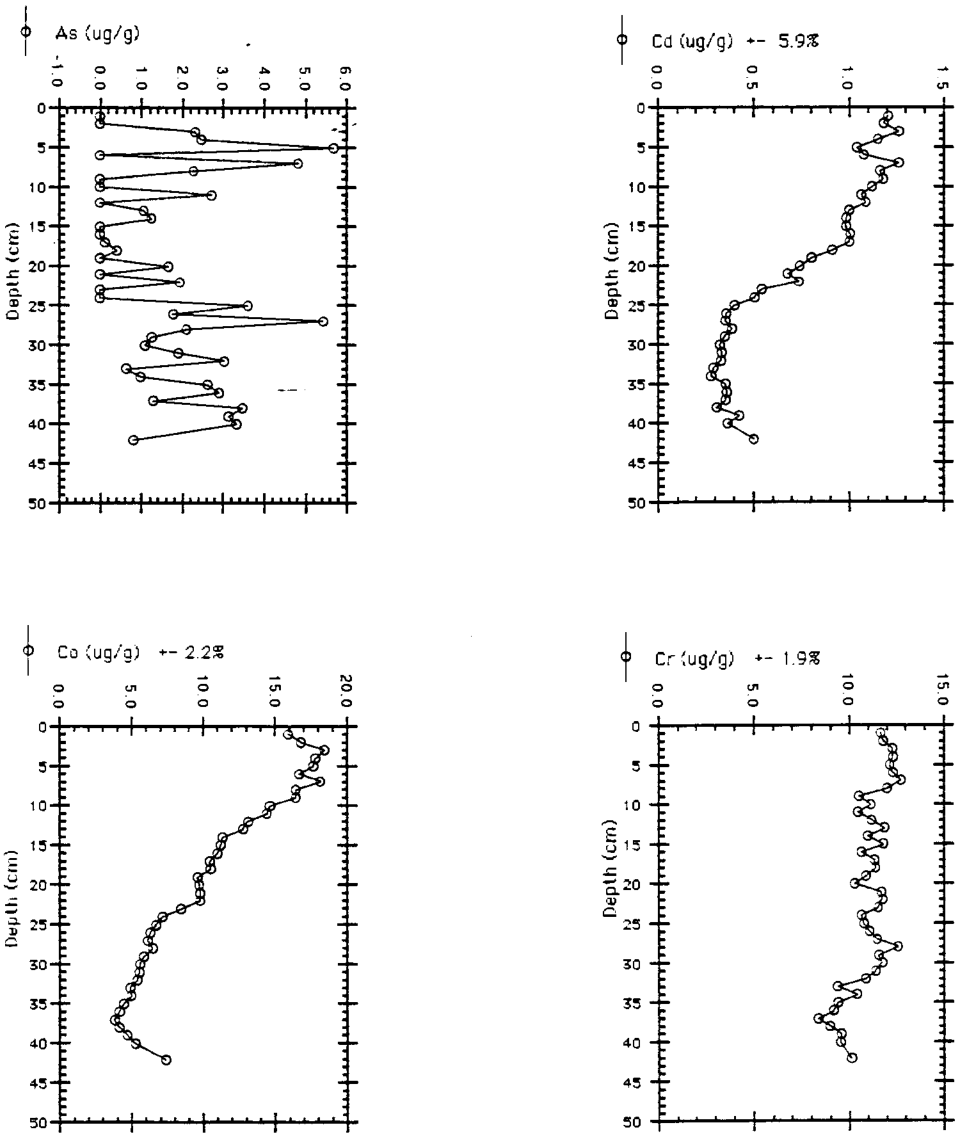


Figure 26: Core B (August, 1990).
Trace Metal Concentration vs Depth.

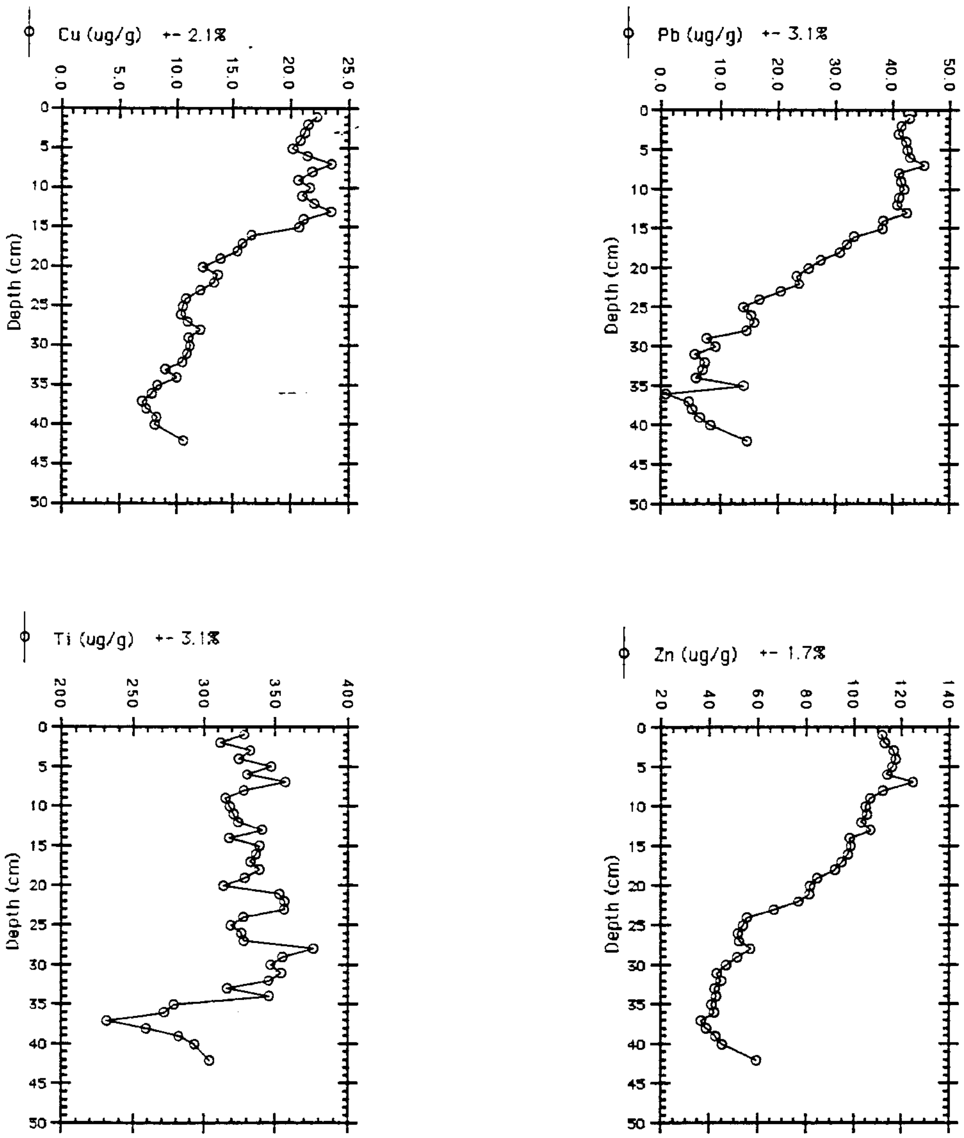


Figure 26: Core B (August, 1990).
Trace Metal Concentration vs Depth.

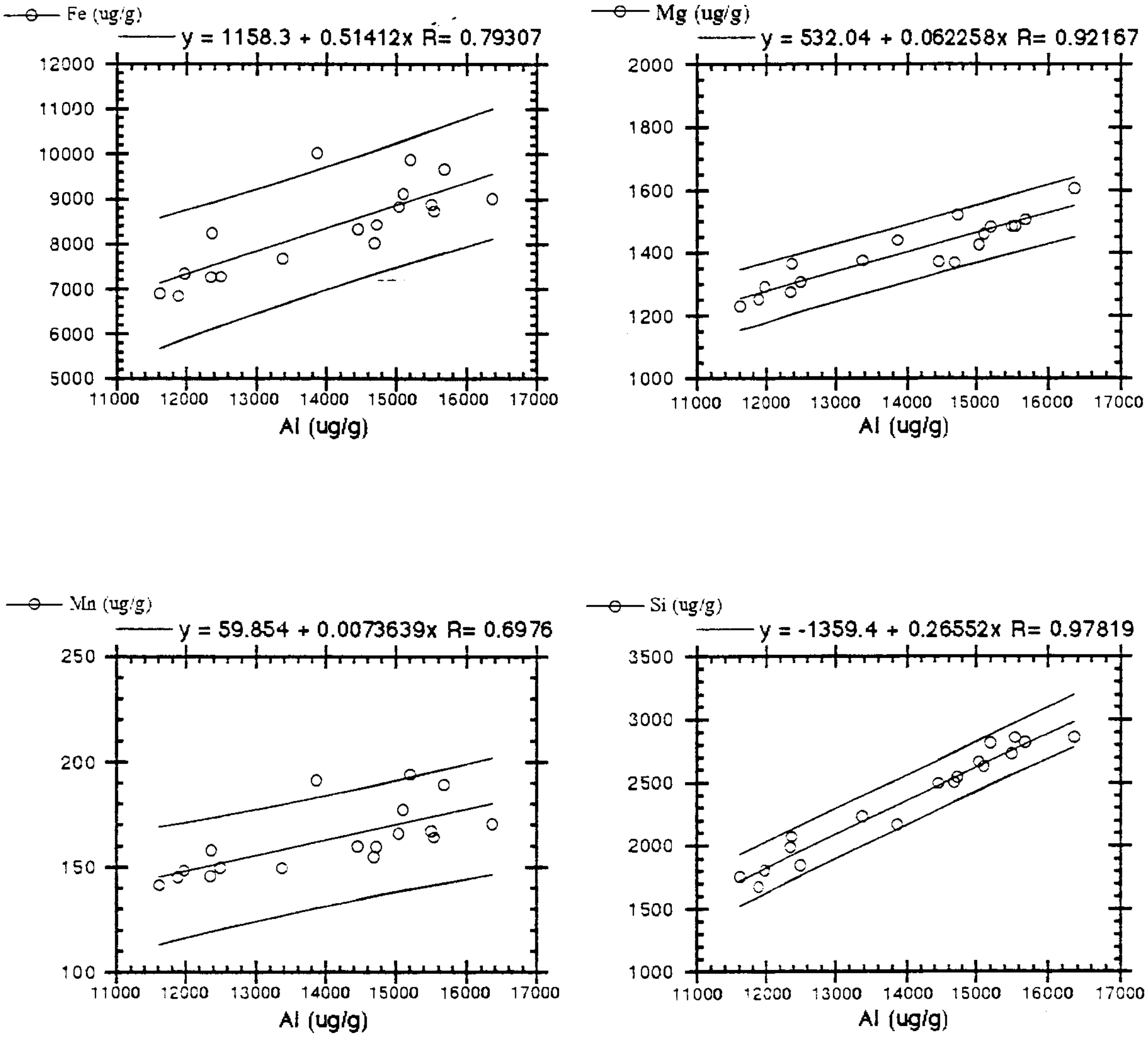


Figure 27: Major metals to Aluminum Concentration from Core B Background sections (24-42 cm). Linear regression lines and +-95% confidence limits are also plotted to represent ambient concentration zones.

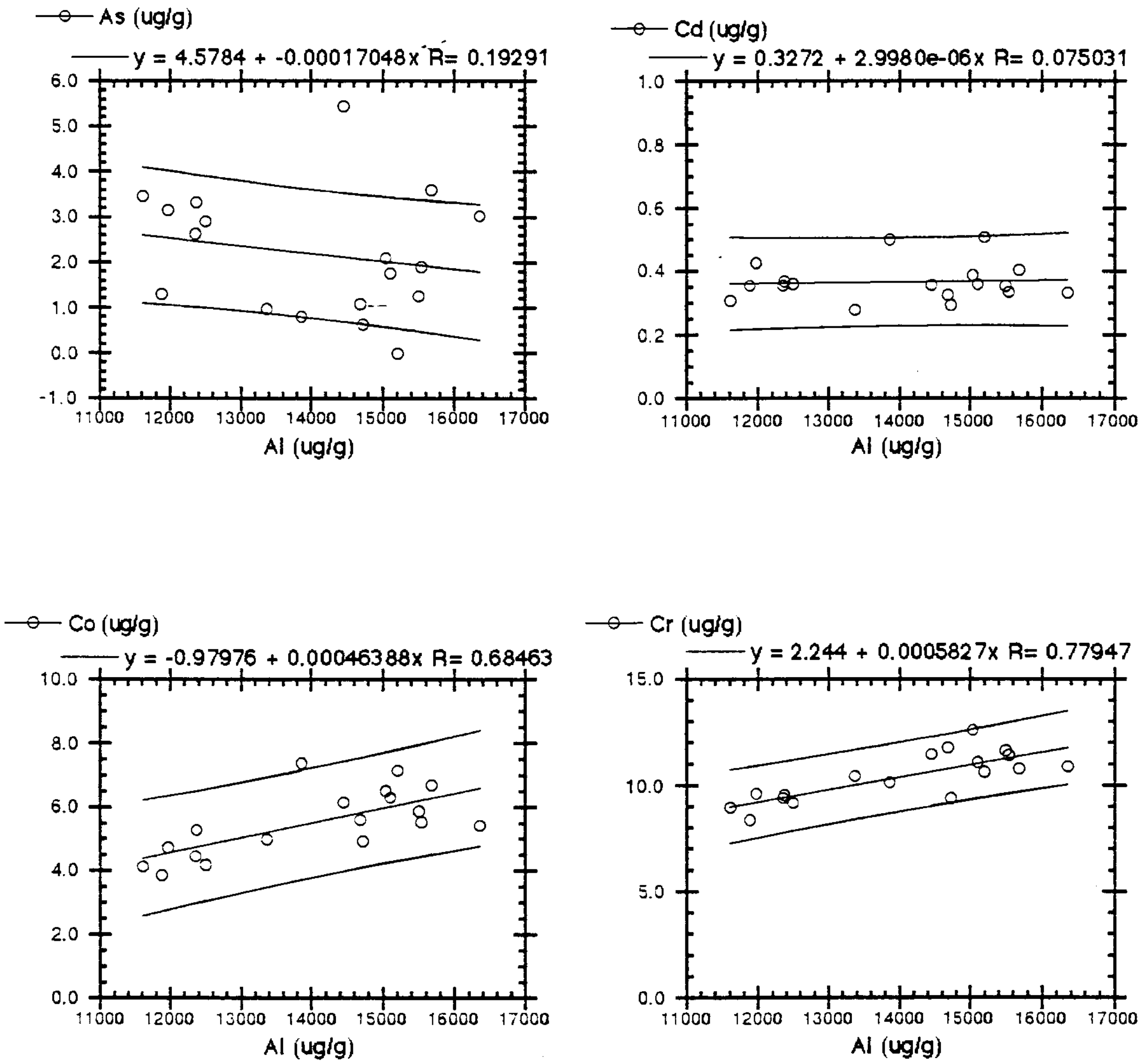


Figure 27: Trace Metals to Aluminum Concentration from Core B Background sections (24-42 cm). Linear regression lines and +-95% confidence limits are also plotted to represent ambient concentration zones.

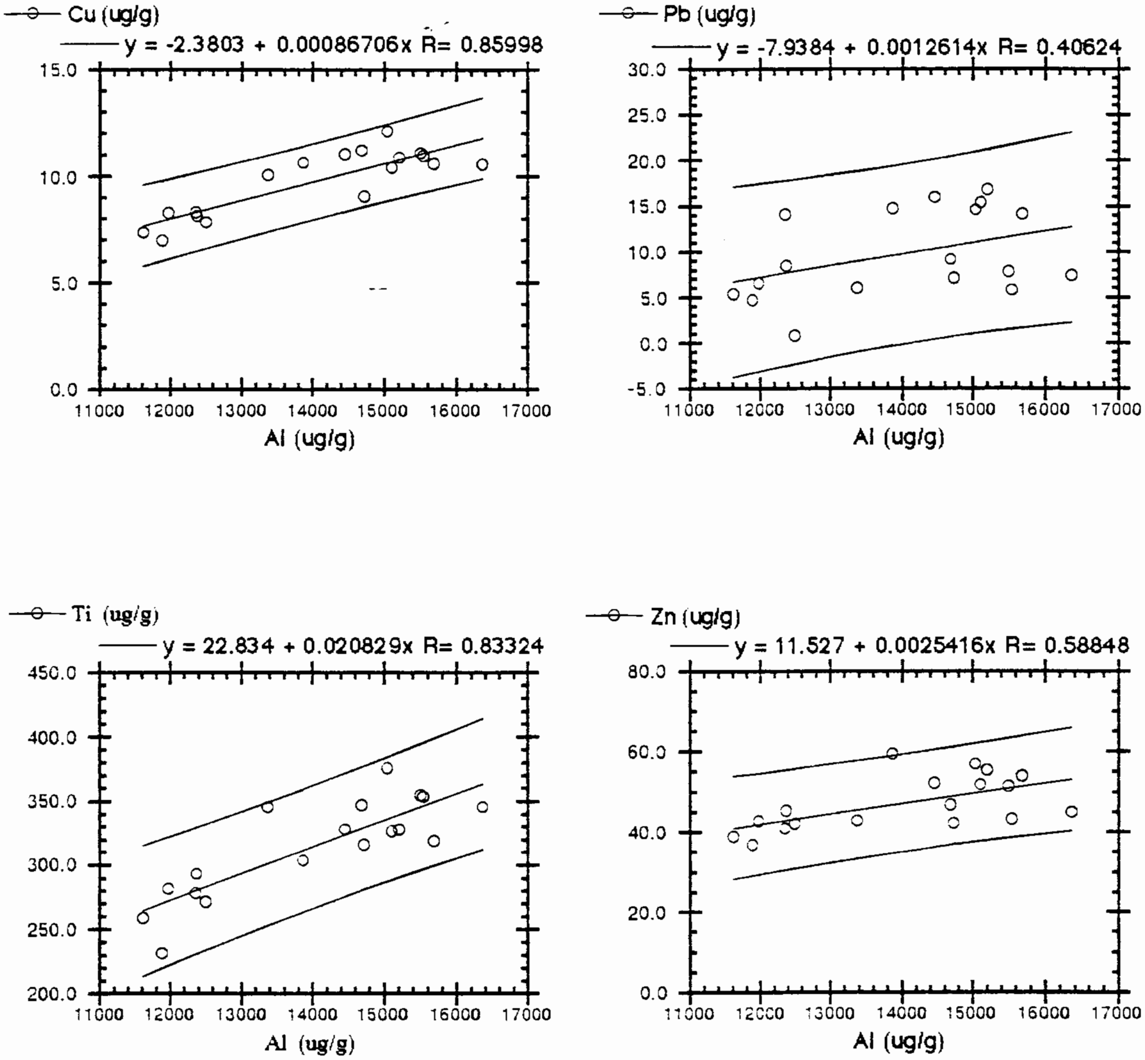


Figure 27: Trace Metals to Aluminum Concentration from Core B Background sections (24-42 cm). Linear regression lines and +-95% confidence limits are also plotted to represent ambient concentration zones.

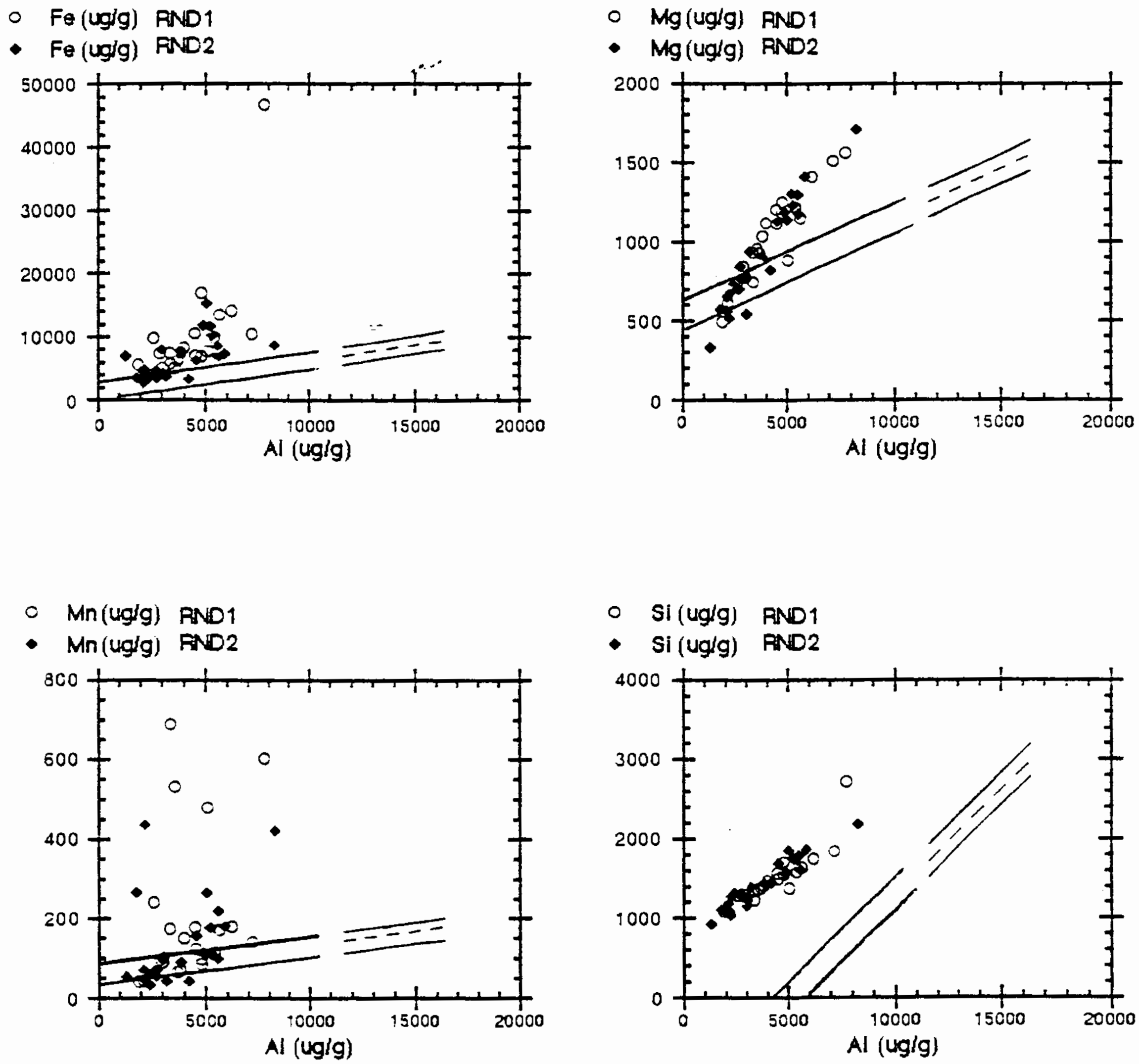


Figure 28: Major Metals to Aluminum Correlation for Frazier Brook Sediment Samples, (June & October, 1990). Regression lines and +-95% confidence limits from Core B background sections.

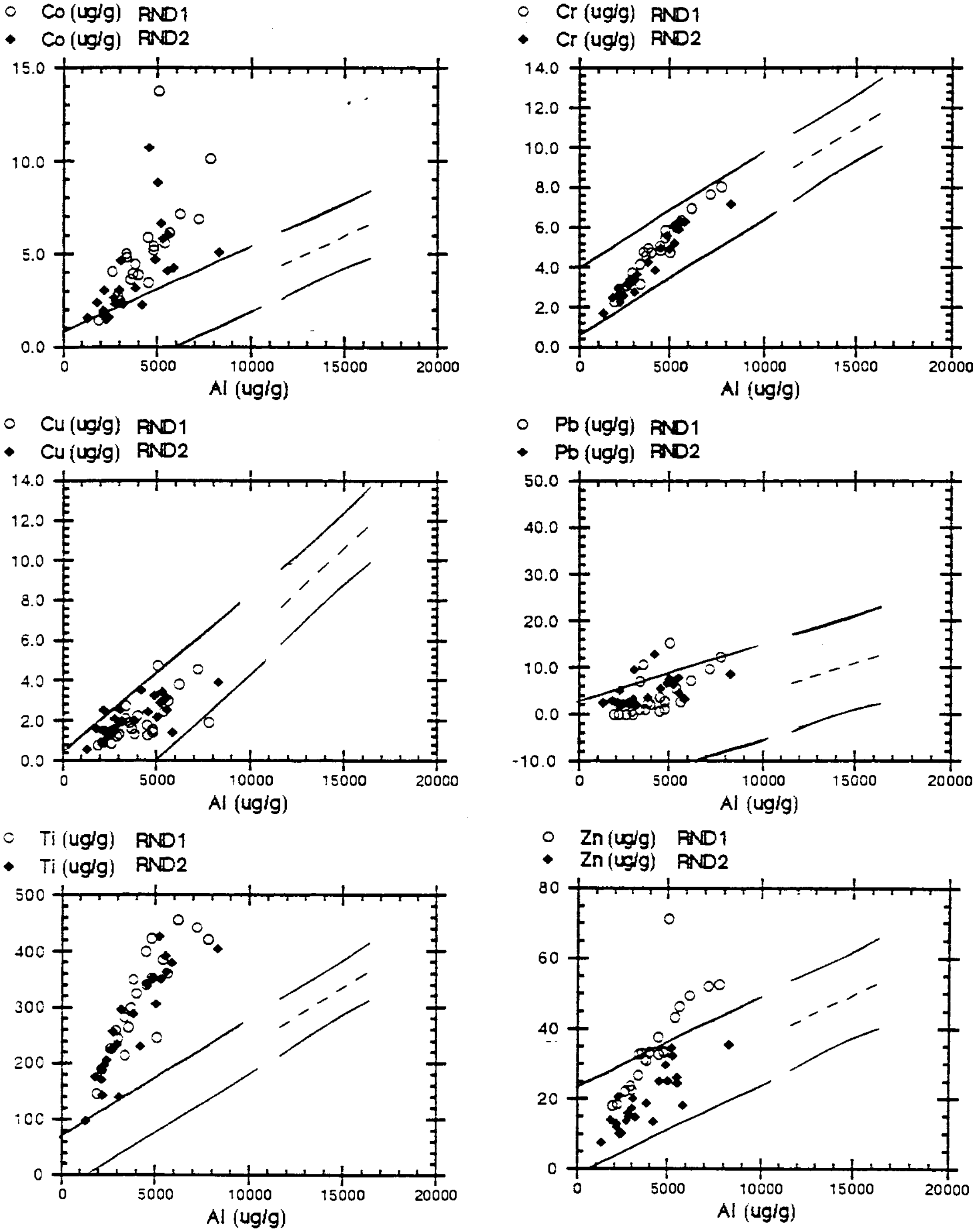


Figure 28: Trace Metals to Aluminum Correlation in Frazier Brook Grab Samples (June vs October,1990). Regression lines and + - 95% confidence limits from Core B background sections.

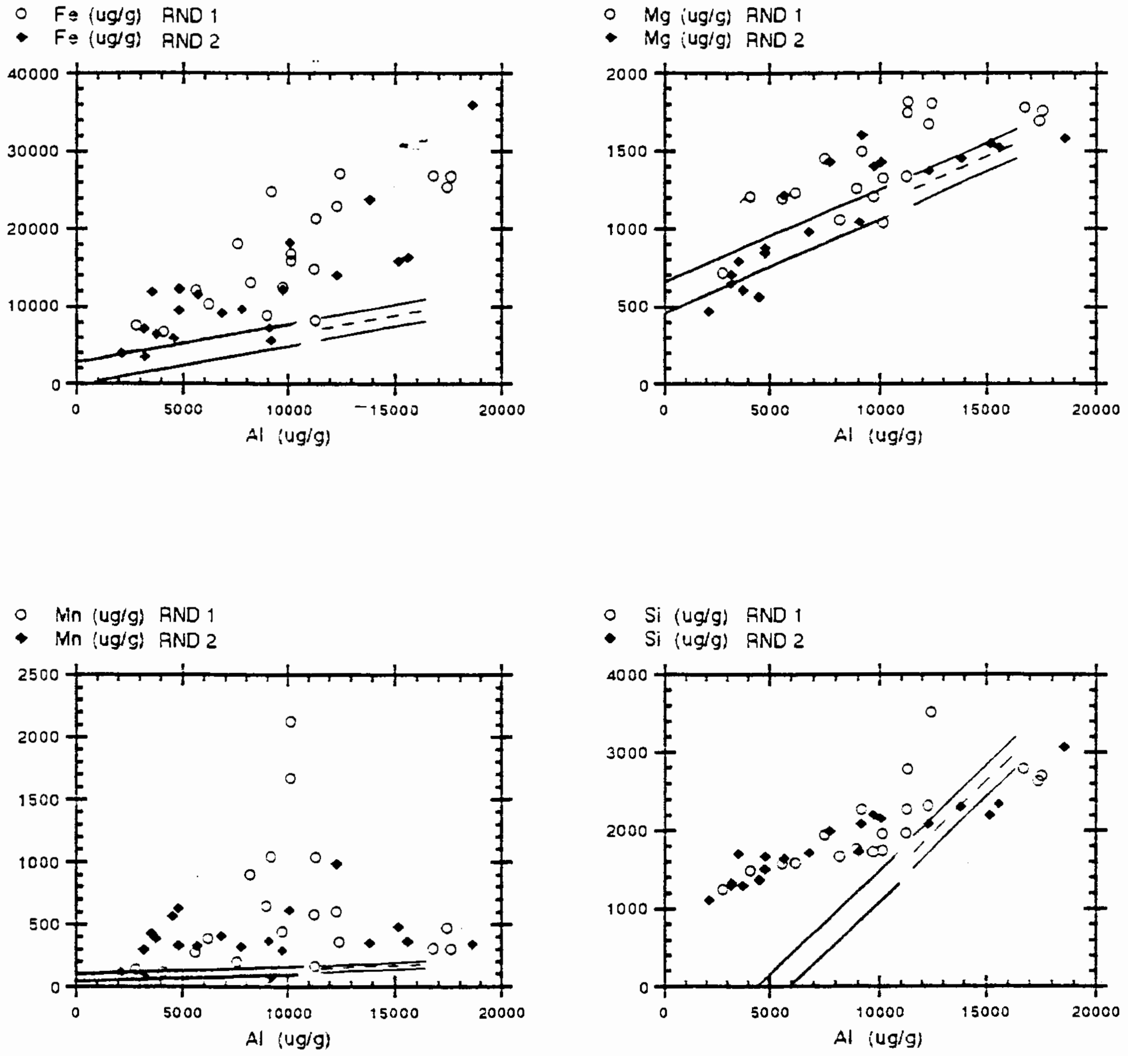


Figure 29: Major Metals to Aluminum Correlation for Eagle Pond Grab Samples (June & October, 1990). Regression lines and +-95% confidence limits from Core B background sections.

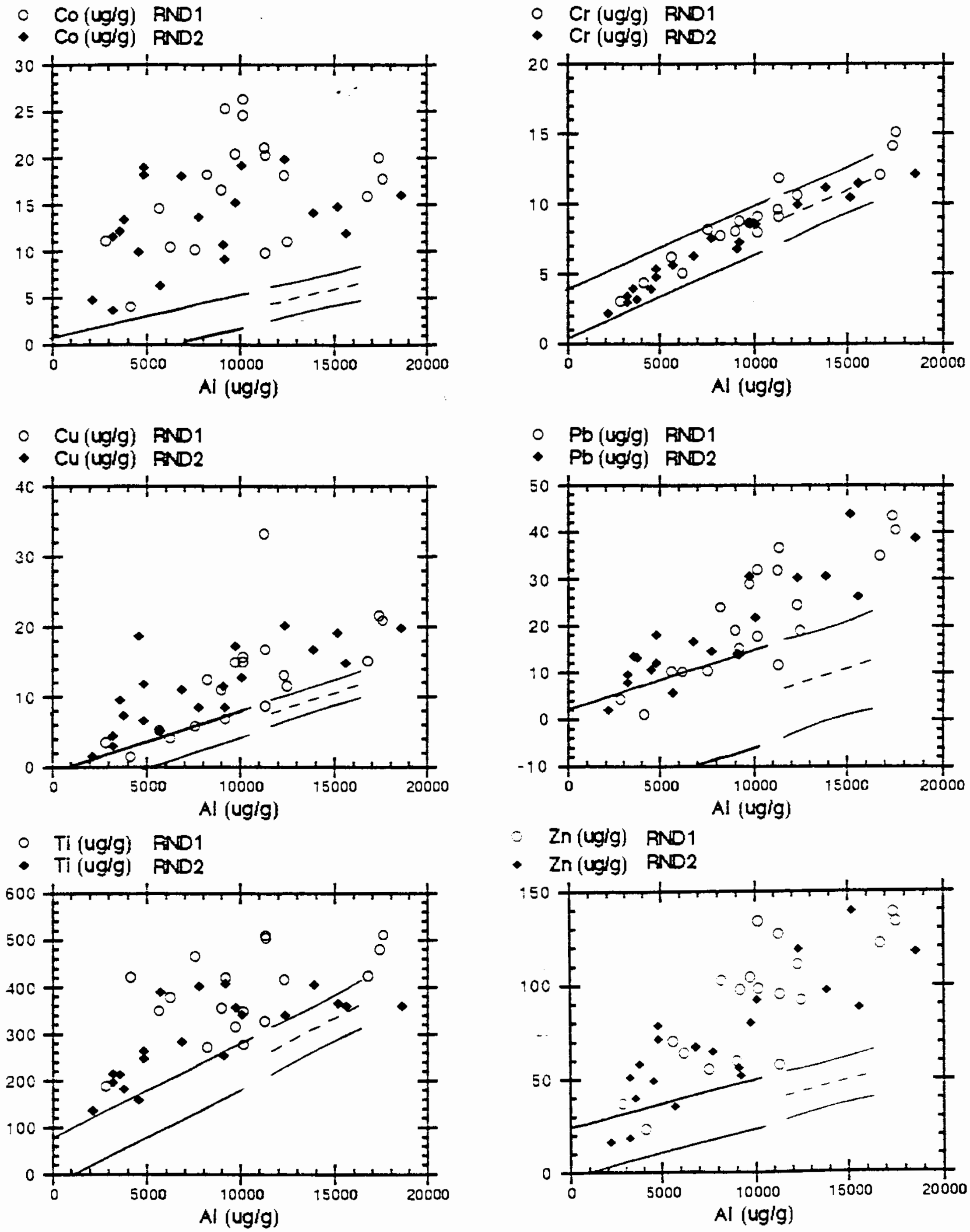


Figure 29: Trace Metals to Aluminum Correlation for Eagle Pond (June & October, 1990). Regression lines and +-95% confidence limits from Core B background sections.

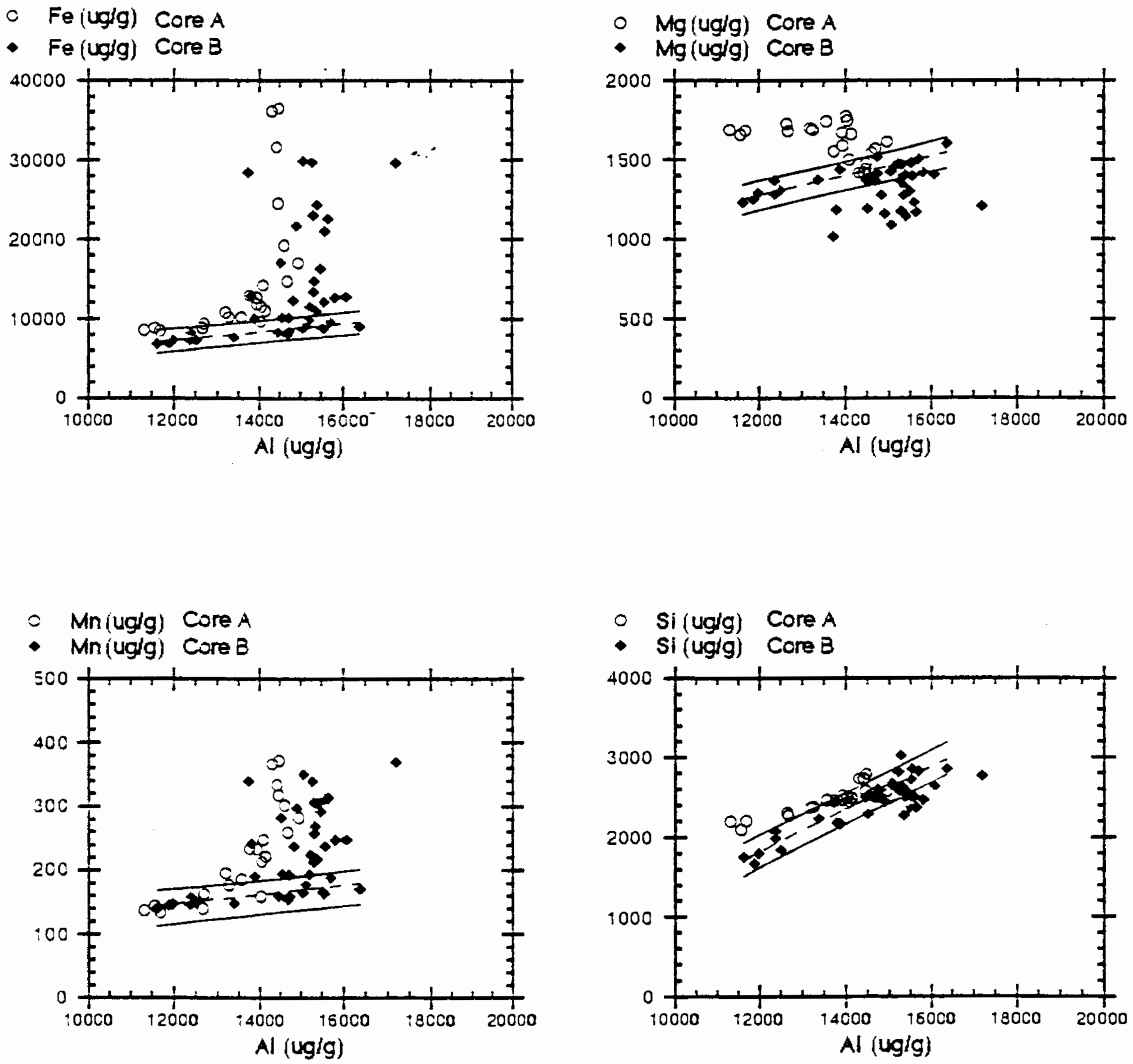


Figure 30: Major Metals to Aluminum Correlation for Eagle Pond Cores A & B. Regression lines and+-95% confidence limits from Core B background sections.

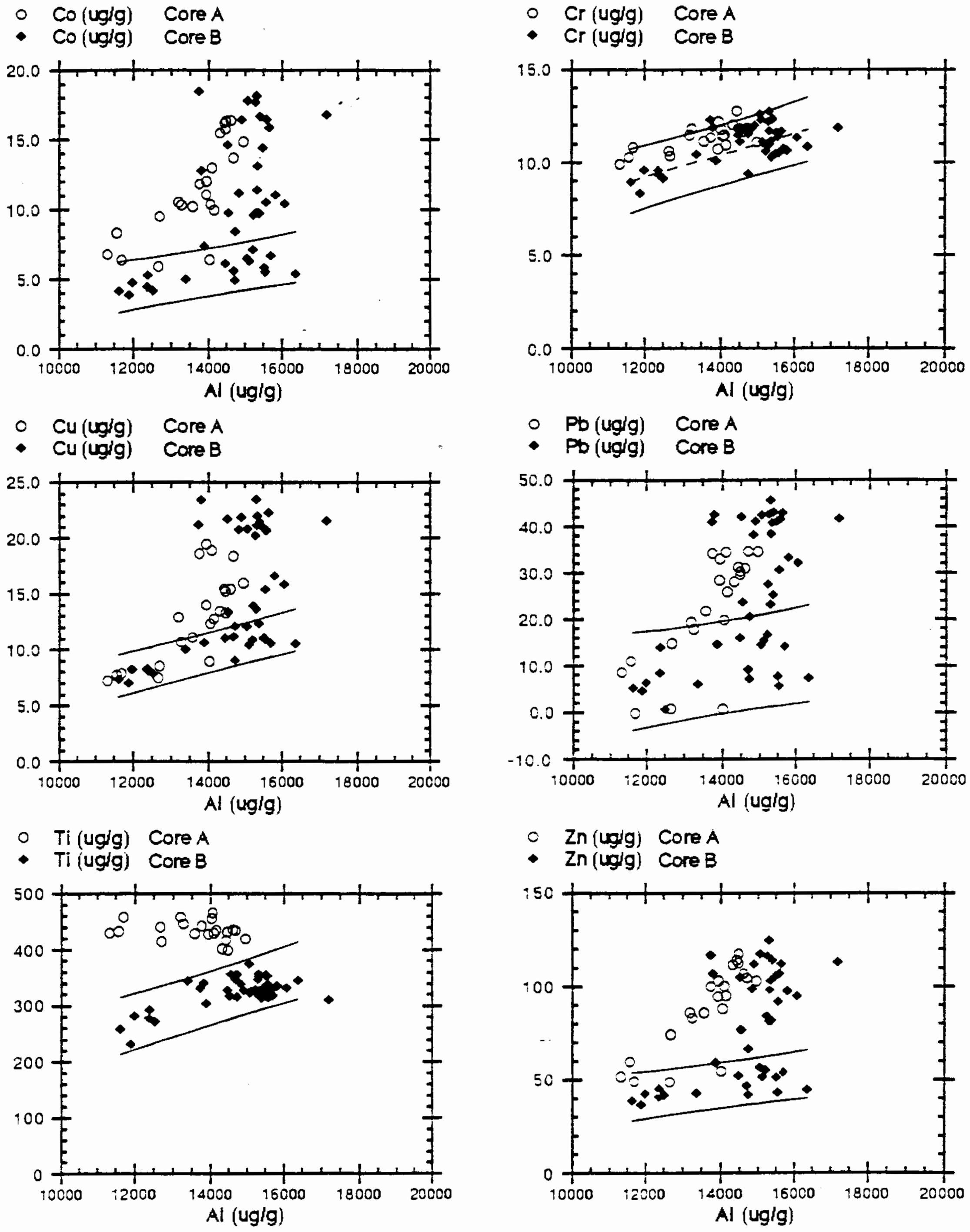


Figure 30: Trace Metals to Aluminum Correlation for Eagle Pond Cores A & B. Regression lines and+-95% confidence limits from Core B background sections.

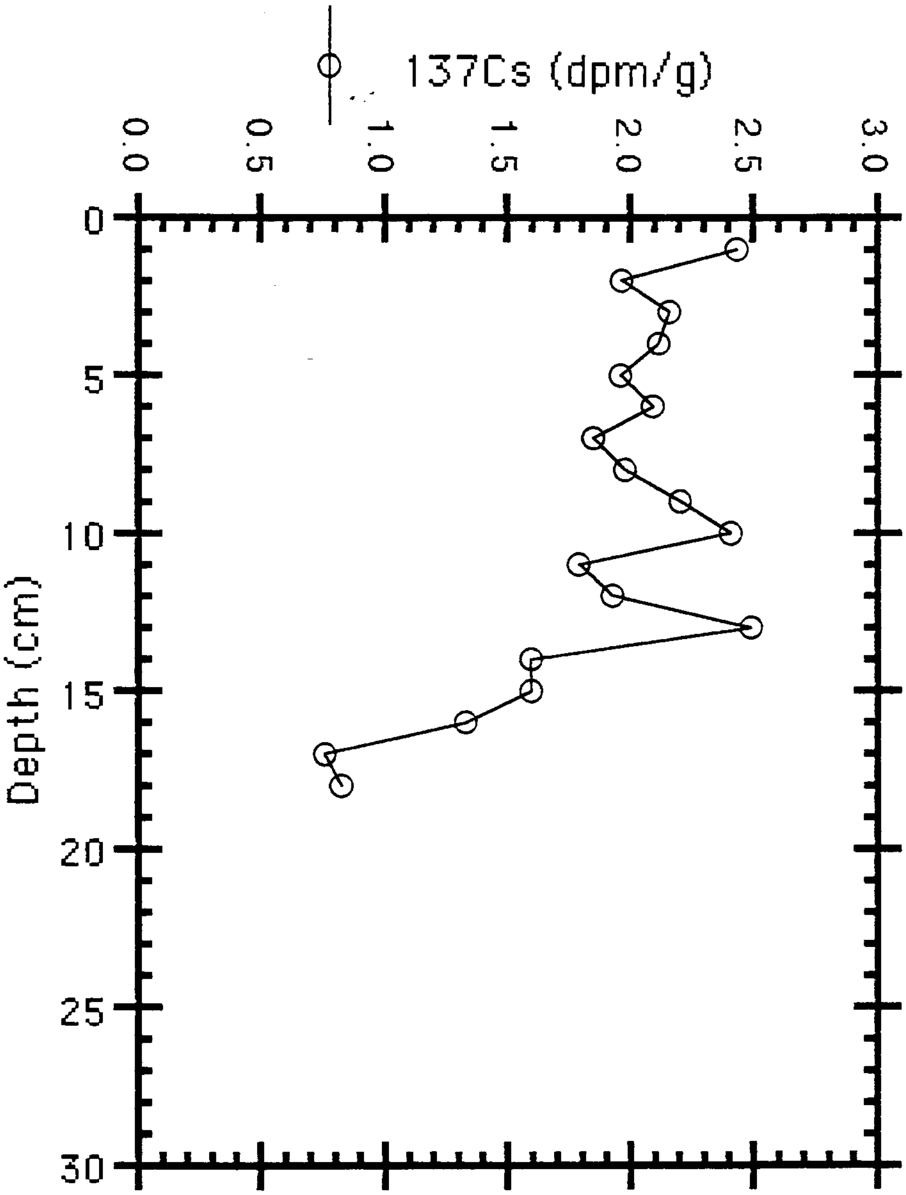


Figure 31 : Core C; Activity of Cs-137 vs. Depth. Sedimentation rate estimated by assuming top of core is 1991 and depth at which Cs-137 is first detected is 1954. Rate, therefore, estimated as 18cm/37yr=0.49cm/yr.

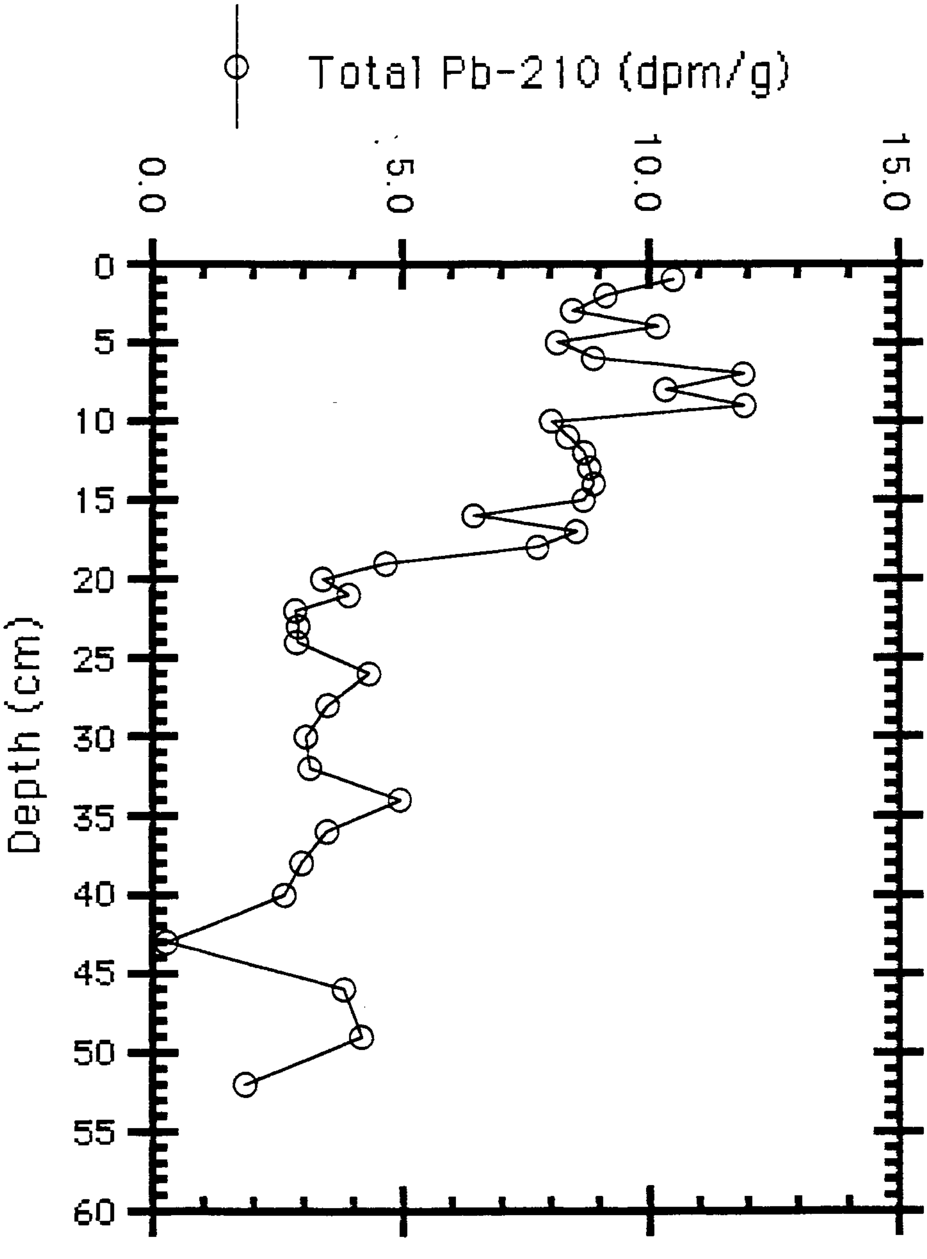


Figure 32: Core C; Total Pb-210 Activity Vs Depth

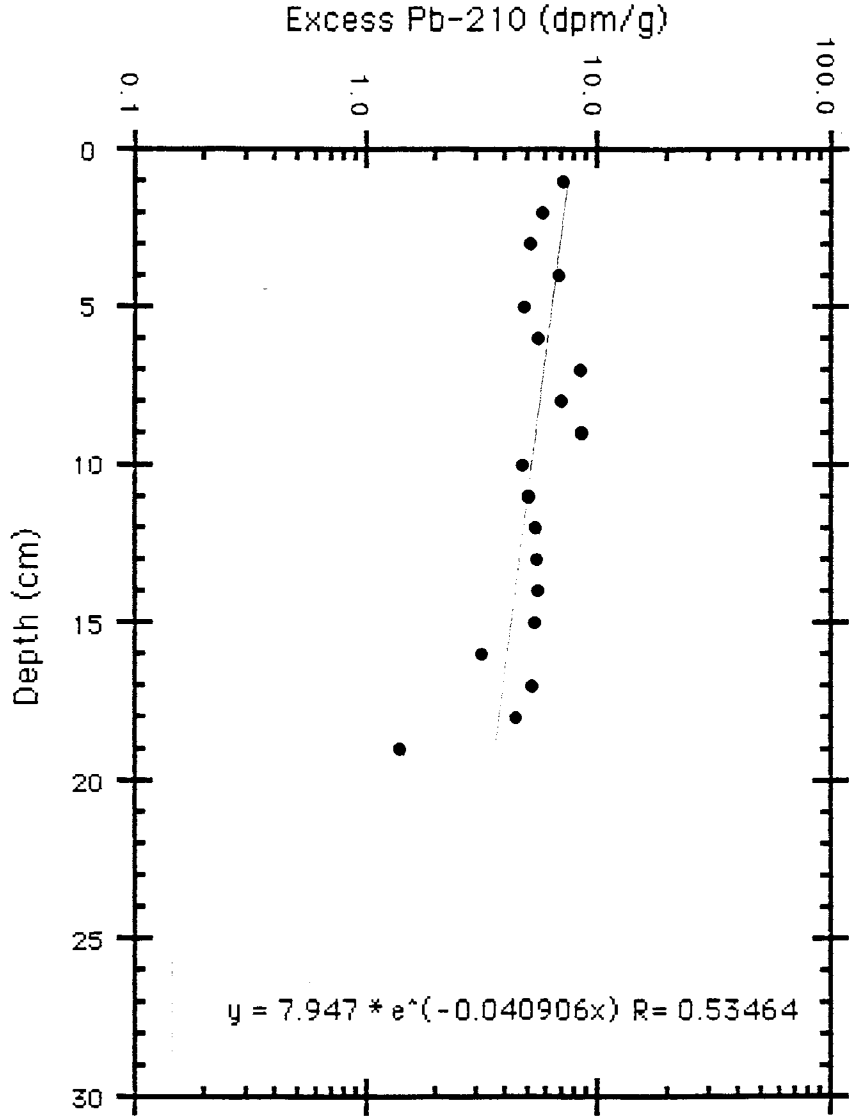


Figure 33: Excess Pb-210 Activity vs Depth in top sections of Core C. Sedimentation rate (a) derived from slope (m) of best fit line: a = -1/m = -0.0311/-0.041 = 0.76 cm/yr.

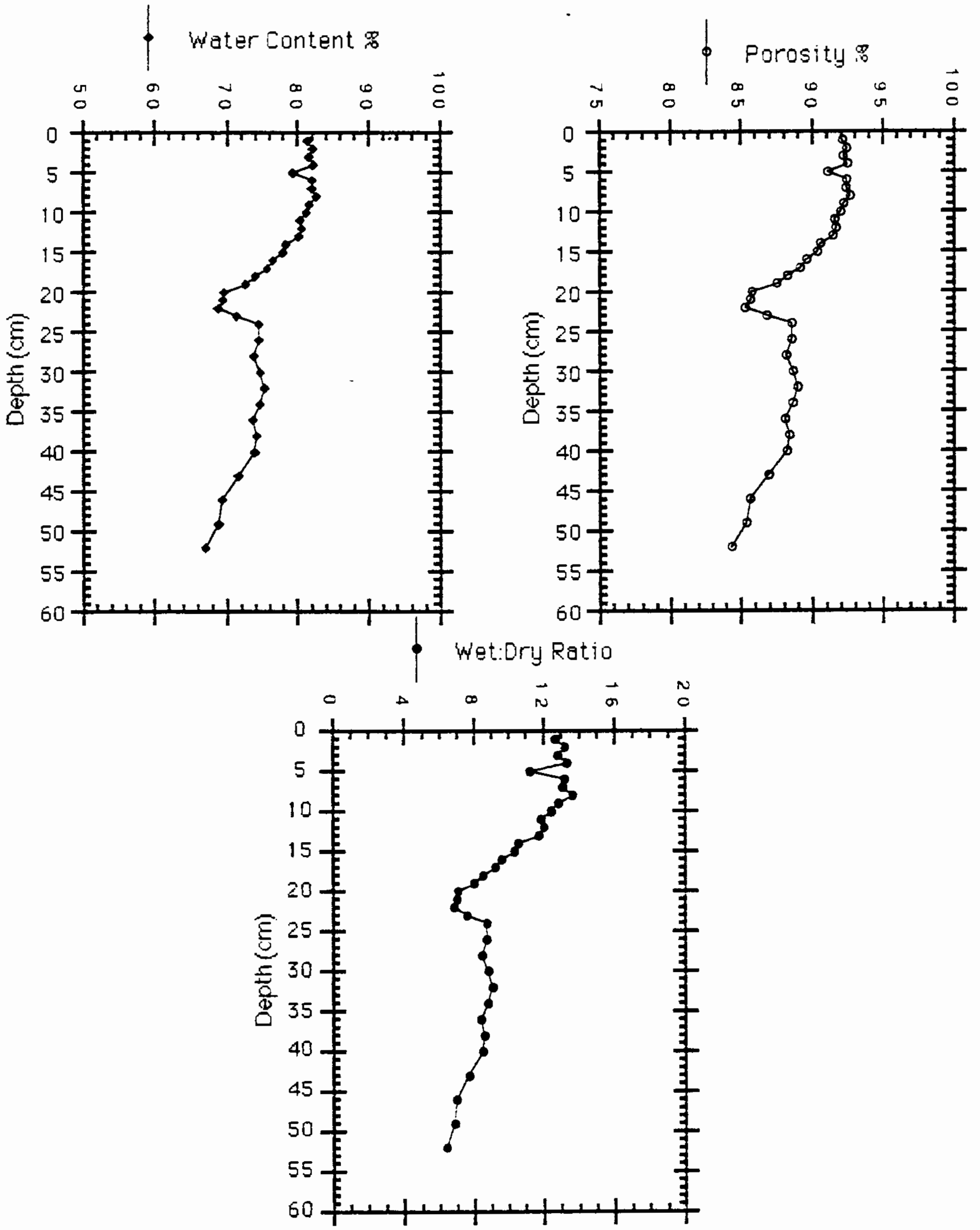


Figure 34: Percent Water Content (mass) determined by: [Mw/(Mw+Ms)]*100. Porosity determined by: [Mw/dw/(Mw/dw)+(Ms/ds)] = Volw/(Volw+Vols)*100 Vol Wet:Dry Ratios from (Volw+VolS)/Vols

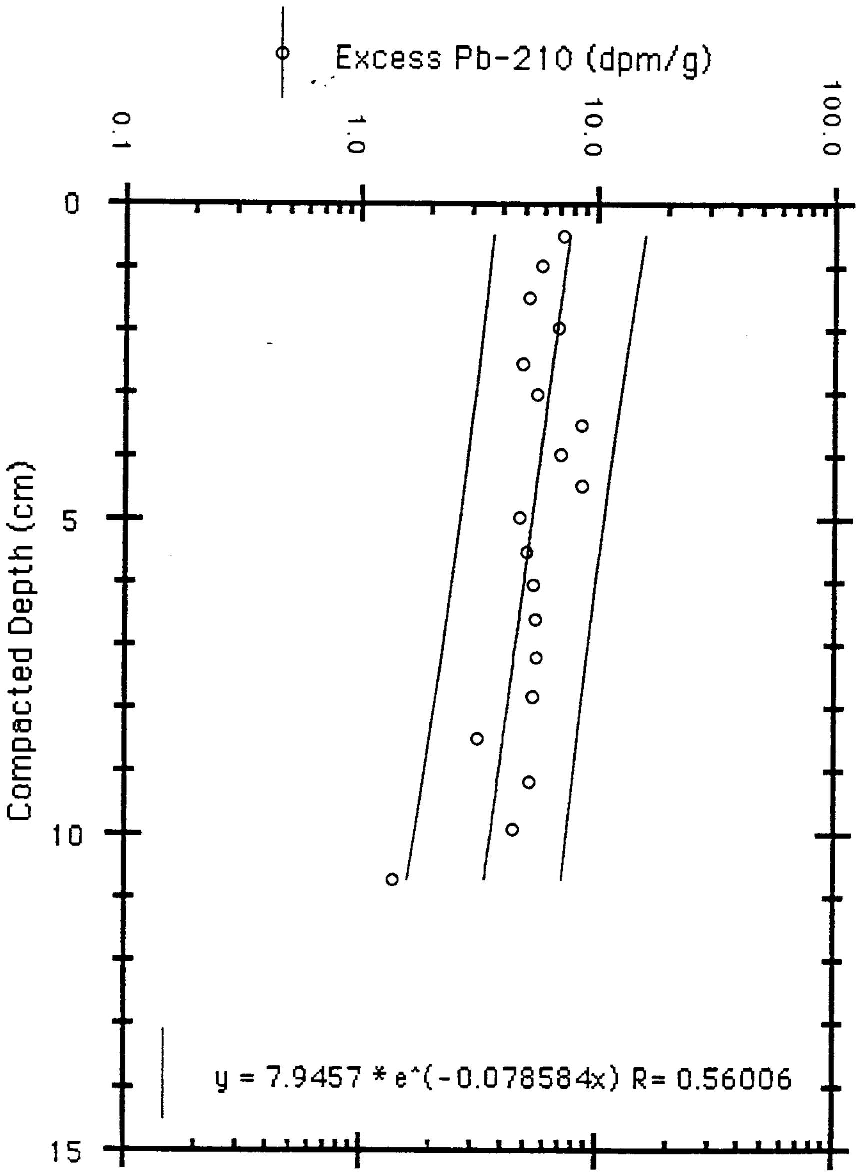


Figure 35: Excess Pb-210 vs Compacted Depth. From slope of best fit line, sedimentation rate (a) = -1/m = -0.0311/-0.079 = 0.39 cm/yr.

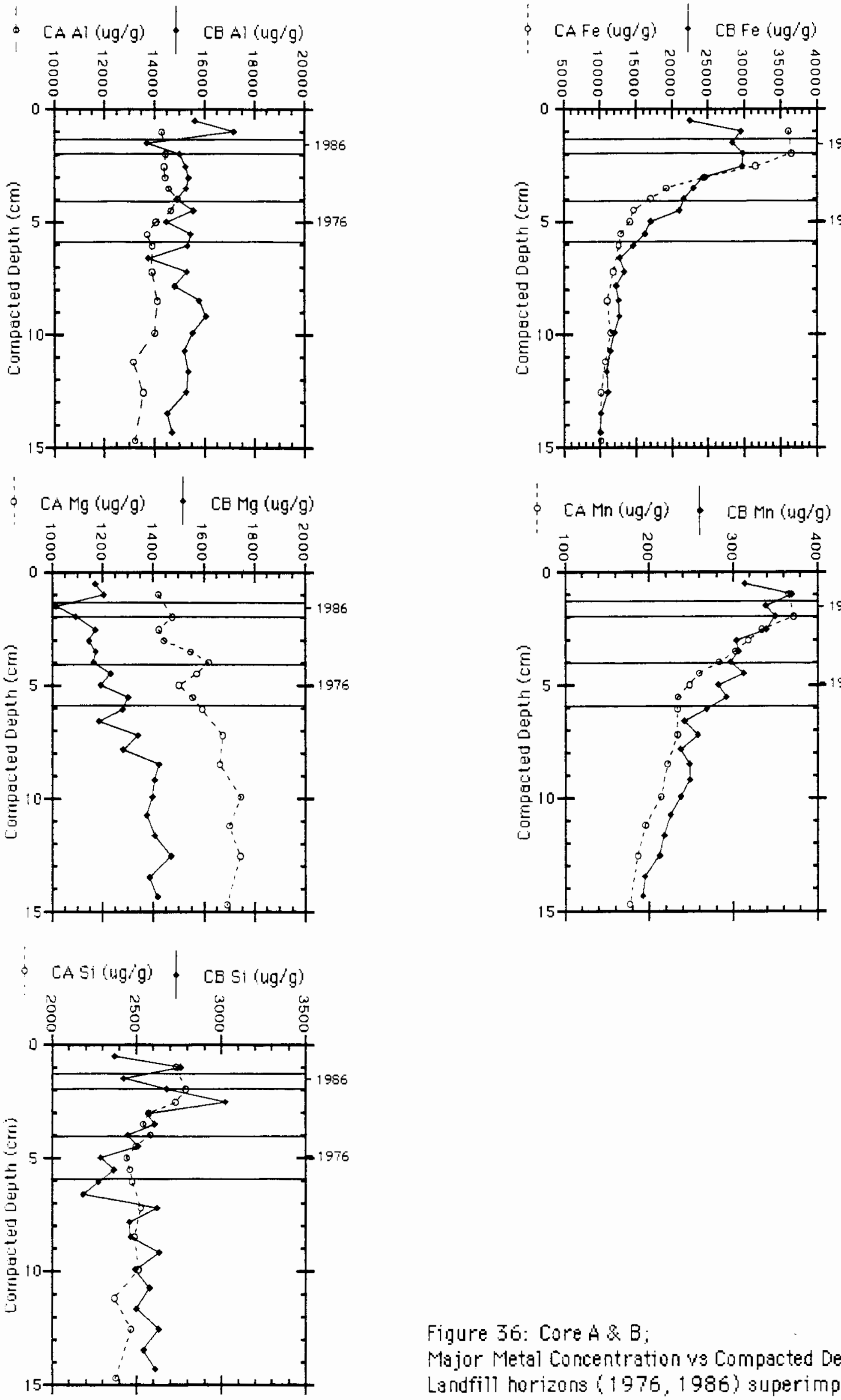


Figure 36: Core A & B;
Major Metal Concentration vs Compacted Depth.
Landfill horizons (1976, 1986) superimposed.

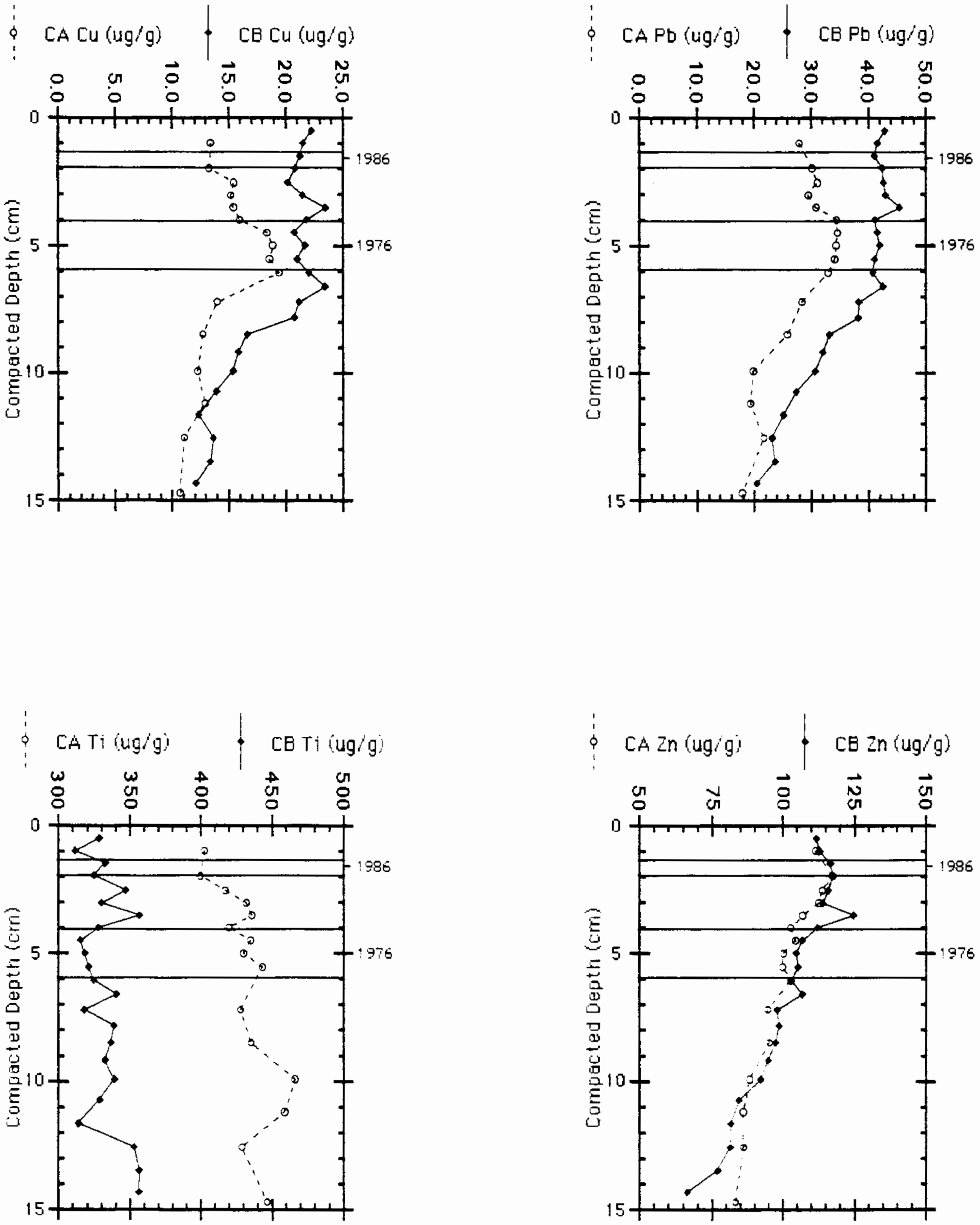


Figure 36: Core A & B;
Major Metal Concentration vs Compacted Depth.
Landfill horizons (1976, 1986) superimposed.

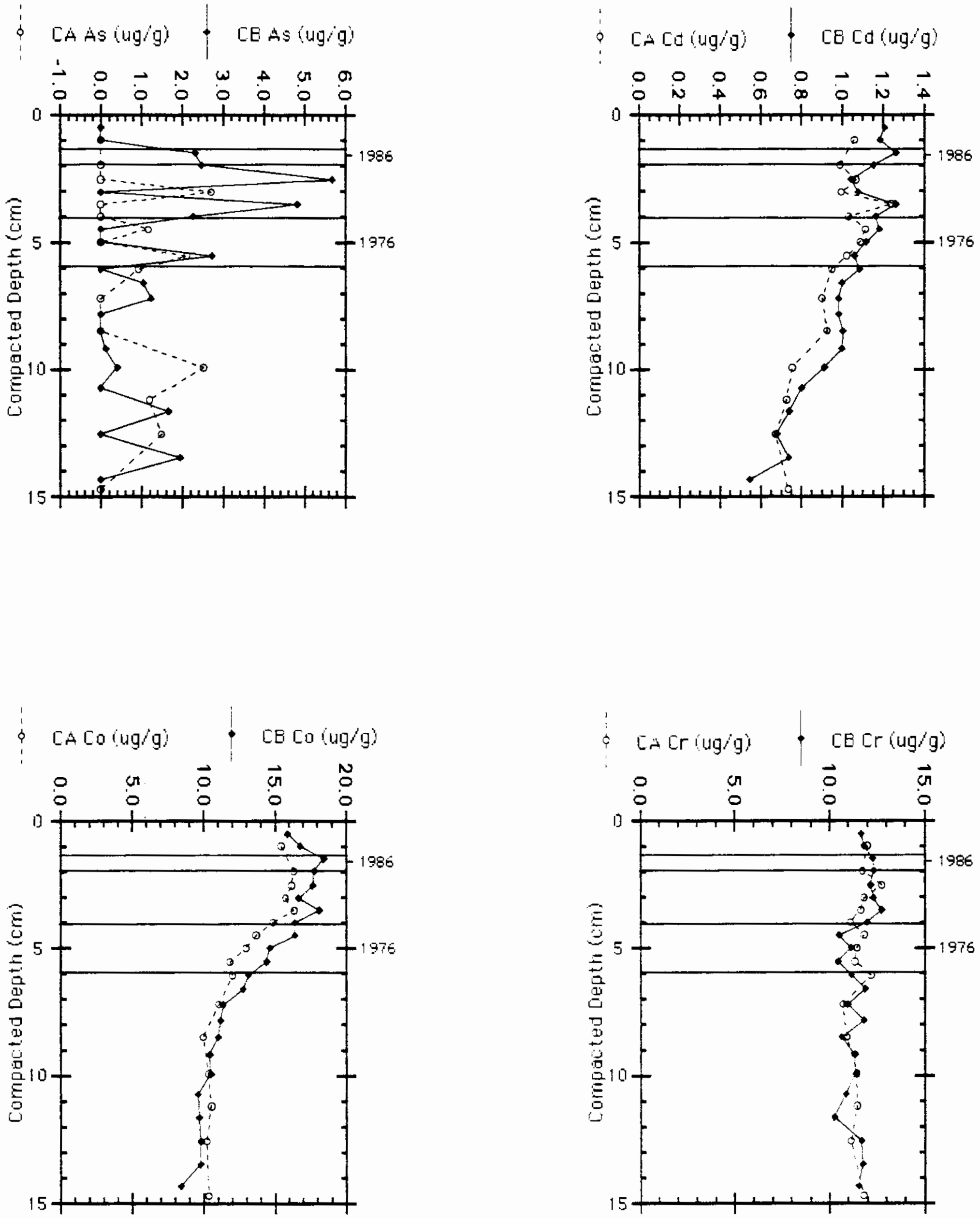


Figure 36: Core A & B;
Major Metal Concentration vs Compacted Depth.
Landfill horizons (1976, 1986) superimposed.