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ABSTRACT

In southwestern British Columbia, a significant portion of the glacial fluvial sand and gravel deposits comprising the Abbotsford Aquifer is contaminated by nitrates exceeding the provincial drinking water quality standard of 10 mg/L as nitrate-nitrogen (NO₃-N). This essentially unconfined aquifer is up to 60 m in thickness and occupies a semi-circular upland area of 50 km² adjacent to the Canada - U.S.A. border. Groundwater flow emanates radially from the central portion of the upland (recharge area) towards its outlying boundaries where discharge is evident along the peripheral toe of the upland. Non-pumping water levels range from near ground surface to 40 m below ground. Water quality data obtained from long-term monitoring of a number of sites over a 15-year period indicate that nitrate concentrations vary seasonally. Highest nitrate concentrations generally occur during the winter months when nitrate is leached to the water table during recharge from infiltrating precipitation and rising water table conditions. An increasing temporal trend in nitrate concentration is generally evident in groundwaters within the eastern discharge area where nitrate-nitrogen levels have risen at one site from 1.2 mg/L in 1974 to 8.3 mg/L in 1984. Site specific determinations of water quality with depth at a control site instrumented with 12 standpipe piezometers indicate that nitrate concentrations decrease with depth. Groundwater high in nitrates correlates significantly with increased concentrations of chloride and specific conductance suggesting an organic source of contaminants such as animal wastes. Absence of tritium in groundwater from domestic wells as shallow as 10 m in depth suggests a relatively long history of nitrate contamination probably originating after

World War II. $\delta^{15}\text{N}$ isotope determinations indicate enrichment of ^{15}N with values of $\delta^{15}\text{N}$ in the range +5.5 to 23.7 ppt. indicative of nitrogen derived from organic sources such as soil humus and animal wastes. These results and observations of land use practices indicate that poultry manure wastes which are inappropriately stored, disposed of and applied to the land are a principle source of nitrate contamination of groundwaters in the area.

INTRODUCTION

Degradation of potable groundwater supplies by non-point sources of pollution is of growing concern in many rural areas of British Columbia. Groundwater quality is especially vulnerable to contamination in the glaciated intermontane valleys underlain by highly permeable sand and gravel aquifers which commonly occur under relatively shallow water-table conditions. Nitrate is a major groundwater contaminant which has been identified in a number of areas including the Fraser Lowland physiographic region located in the southwestern portion of the Province (Figure 1). One specific aquifer which has been affected is the Abbotsford Aquifer which occupies a local, semi-circular upland area (Abbotsford Upland) of 50 square kilometres (19 square miles) near the community of Abbotsford (Figure 2). This essentially unconfined aquifer is an important source of water supply for local municipal, irrigation, industrial and domestic requirements (Kohut, 1987).

This paper presents some of the preliminary findings of the water quality investigations carried out by the Water Management Branch of the British Columbia Ministry of Environment on the extent, nature and probable sources of nitrate

contamination in the Abbotsford Aquifer. Investigations were initiated in 1985 and groundwater quality monitoring is continuing. Previous groundwater quality monitoring investigations carried out since 1978 and reported by Halstead (1986) indicate that shallow dug wells within the water table aquifers of the Fraser Lowland are particularly susceptible to contamination by leachate from common stockpiles of animal wastes, especially poultry manure.

STUDY AREA

The Abbotsford Upland lies between elevations of 45 to 75 metres (150 to 250 feet) above sea level. The area is bounded on the south by the International Boundary, on the west by Fishtrap Creek, on the north by the Matsqui Prairie Lowland, and on the east by the Sumas Prairie Lowland (Figure 2). The eastern boundary is marked by an escarpment which rises abruptly from the Sumas Prairie Lowland which is at an elevation of 12 m (40 feet) above sea level. There are no major streams dissecting the Upland, but a number of small lakes occur on the Upland, the surfaces of which are believed to be an expression of the water table (Halstead, 1959). A line of springs (discharge area) occurs along the eastern toe of the Upland.

The region is characterized by a cool Mediterranean type of climate in which precipitation falls principally as rainfall during the period September to May. The area receives an average of 1513 mm (60 inches) of precipitation annually (Environment Canada, 198_). The period from June to September is normally dry and may be subject to drought conditions.

Land use on the Upland is predominantly agricultural with associated local food-processing/producing industries. The Upland is underlain by sandy,

coarse-textured and well-drained soils developed on glacial-fluvial deposits making conditions favourable for the growing of berry crops including strawberries and raspberries. Despite the relatively high annual precipitation, irrigation is required during the summer months. Poultry farming is also a major agricultural activity on the Upland area resulting in widespread stockpiling and application of poultry manure litter to the land. The litter which is high in organic content is of particular value as a soil conditioner; improving the moisture retention properties of the sandy deposits.

HYDROGEOLOGY

The Abbotsford Upland is underlain by a succession of unconsolidated glacial and non-glacial deposits of Pleistocene and Recent ages. These deposits are reported to be at least 103 m (339 feet) in thickness at one locale where they overlie bedrock of Tertiary age (Armstrong, 1960). Recessional outwash and advance outwash deposits of sand and gravel generally comprise the surface deposits to a depth of 60 m (200 feet). These are interspersed with discontinuous bodies of till. These materials comprise a major hydrostratigraphic unit which has been termed the Abbotsford Aquifer (Kohut et al, 1982). Transmissivity values in the range 1.4×10^{-2} to 2.2×10^{-2} m^2/S (1.0×10^5 x 1.5×10^5 USgpd/ft. width of aquifer) have been reported along the eastern toe of the Upland (Callan, 1971). Groundwater generally occurs under non-confined or water-table conditions with water levels ranging from 0 to 40 m (0 to 130 feet) below ground. Water levels in the western portion of the Upland are generally shallow (<6 m [20 feet] below ground) as shown in Figure 3. The deepest water levels are found in the central

and eastern, topographically higher areas of the Upland. Regional groundwater flow determined from historic non-pumping water levels obtained when the wells were first drilled or inventoried (Kohut, 1987), is radially away from the topographically higher north-central portion of the Upland towards the outlying western, eastern and northern boundaries. Groundwater flow in the southern portion of the Upland appears to be towards the State of Washington, U.S.A.

METHODS OF INVESTIGATION AND RESULTS

Three stages of water quality investigation have been carried out to date.

These were:

1. a review of available historical water quality data on file for the area from various sources including chemical analyses undertaken for Environment Canada, Inland Waters Directorate and the Provincial Ministries of Health and of Environment;
2. periodic field sampling of domestic wells and observation wells for field and laboratory determination of nitrates;
3. establishment of a control site equipped with 12 nested piezometers to sample nitrate variations with depth, determine hydraulic gradients, direction of groundwater flow, nitrate movement and probable sources of contamination.

A brief description of each stage of investigation and results obtained are given below.

Review of Historic Water Quality Data

Following a review of existing groundwater quality information for three aquifer regions in the Fraser Lowland, Kwong (1986a) identified 2 areas on the Abbotsford Upland with groundwaters having concentrations of $\text{NO}_3\text{-N}$ exceeding the provincial drinking water quality standard of 10 mg/L (Ministry of Health, 1983). Drinking waters having nitrates in excess of 10 mg/L as nitrogen have been linked to the occurrence of infantile methemoglobinemia (Health and Welfare Canada, 1979). In this condition the oxygen-carrying capacity of blood is reduced which can result in death.

Out of 50 domestic well sites having nitrate data, 23 sites (46 percent) showed $\text{NO}_3\text{-N}$ greater than 10 mg/L including 6 sites over 20 mg/L. The extent of the apparent $\text{NO}_3\text{-N}$ contamination is shown diagrammatically in Figure 2 where the concentrations have been contoured. Kwong (1986a) also reported on monitoring results from two well sites (9023N and 9024N, Figure 2) which were sampled periodically by Environment Canada between 1974 and 1979. Site 9023N, a 24 m (81 foot) deep well with a non-pumping water level near 15 m (50 feet) showed a decreasing nitrate concentration from 17 to 13 mg/L between 1976 and 1979. Site 9024N, a 50 m (163 foot) deep well with a non-pumping water level near 27 m (90 feet) also showed a decreasing trend from 10 to 7 mg/L between 1975 and 1979. The temporal data also exhibited seasonal fluctuations with higher nitrate concentrations generally occurring during the fall and winter months when infiltration from precipitation results in rising water-table conditions. Groundwater from two large capacity 126 L/s (2000 USgpm) production wells situated near the Fraser Valley Trout Hatchery and within the discharge area along the eastern toe of the Upland (Figure 2) have shown an increasing nitrate

concentration with time. $\text{NO}_3\text{-N}$ from one well rose from 2.1 to 6.4 mg/L between 1980 and 1986 while $\text{NO}_3\text{-N}$ in the other well showed an increase from 1.2 to 8.3 mg/L between 1974 and 1984. The widespread occurrence of nitrate contamination exceeding the water quality standard and rising trends in some areas with time warranted further investigations.

Field Sampling of Existing Wells

In March 1986, field sampling of 19 wells was carried out to confirm the findings of the Kwong (1986a) assessment and to obtain up-to-date information on the degree and extent of nitrate contamination. Samples were collected from household taps of domestic water systems connected to wells and from 6 existing Ministry of Environment observation wells on the Upland. Observation wells were sampled with a tube sampler but were not purged before sampling. Laboratory analyses were undertaken by the Environmental Laboratory of the Ministry of Environment for major ions/parameters (Specific Conductance, TDS, alkalinity, sulphate, chloride, potassium, sodium and pH), nitrogen species ($[\text{NO}_3+\text{NO}_2]\text{-N}$, total Kjeldahl-N, $\text{NH}_3\text{-N}$, $\text{NO}_3\text{-N}$, $\text{NO}_2\text{-N}$, ORG-N, total-N) and total and dissolved phosphorous.

Results (Kwong, 1986b) confirmed the extent of the contamination and indicated that concentrations at some sites were increasing while levels at other sites were decreasing. Site 9023N for example, now exhibited an increasing trend from 13 mg/L in 1979 to 19.8 mg/L in 1986. These variable temporal trends were attributed to seasonal variations and possible changes in land use activities in the area (Kwong, 1986b). $\text{NO}_3\text{-N}$ concentrations in the observation wells were anomalously low in the range of <0.02 to 0.26 mg/L while ammonia-N concentrations were surprisingly high in the range of 1.25 to 4.74 mg/L. Although analyses are

reported as ammonia (NH_3) the most likely species in solution is ammonium (NH_4^+) in the in-situ pH (7.5 to 8.5) and temperature ranges (9.5 to 11 degrees C) for groundwater. The absence of nitrate in groundwater from the observation wells has been attributed to localized denitrification by microbial activity in the immediate vicinity of the well and insufficient purging of stagnant borehole water before sampling. Neighbouring production wells showed nitrate to be the dominant species present in the aquifer.

In July 1987, 16 domestic wells having high nitrates were resampled for more complete chemical analyses including 8 sites for dissolved heavy metal determinations, 14 sites for trihalomethane scans, 4 sites for $\delta^{15}\text{N}$ and tritium isotope analyses. Samples for metals were field-filtered through a 0.45 micron cellulose acetate membrane filter and acidified with concentrated nitric acid (2 ml per 500 ml of sample). Major-ion, metals and trihalomethane analyses were undertaken by the Environmental Laboratory of the Ministry of Environment in Vancouver. Samples for $\delta^{15}\text{N}$ analysis were field filtered, packed in a cooler chest with ice packs and sent by courier within 24 hours to the Agriculture Canada, Research Station in Lethbridge, Alberta for analysis. Samples for Tritium analysis were forwarded to the Isotope Laboratory at the University of Waterloo.

Results of the July 1987 sampling, reported by Sather (1988), showed no anomalous heavy metal concentrations. The trihalomethane scans also showed negative results below the detectable limit of 0.001 mg/L suggesting that volatile organic compounds were not prevalent in the groundwaters sampled. $\delta^{15}\text{N}$ determinations were in the range of +3.9 to +9.8 indicative of ^{15}N enrichment and suggesting animal wastes and/or natural soil nitrogen (Kreitler, 1975) as sources of nitrate contamination. Further discussion on the significance of $\delta^{15}\text{N}$ is

provided in the following section on piezometer results. Tritium results for wells as shallow as 10 m in depth and having NO₃-N concentrations as high as 23 mg/L showed no significant concentrations of tritium suggesting a relatively long history of nitrate contamination probably originating after World War II. More detailed site investigations were warranted to ascertain nitrate variations with depth and probable sources of contamination.

Establishment of Control Site with Piezometer Installations

In March 1988, twelve, 50 mm (2-inch) internal diameter standpipe piezometers were installed south of the Abbotsford Airport (Figure 3) in a triangular array with each location (A, B and C) of the array having four piezometers set to different depths up to 23 m (75 feet) below ground (Sather, 1989). Piezometers settings and construction details are shown in Figure 4. Screened intakes were positioned at approximate depths of 23, 17, 11 m (75, 55, 35 feet) and straddling the water table at 5 to 6 m (15 to 20 feet) below ground. Piezometers were installed in 150 mm (6-inch) internal diameter steel cased holes (2 per hole) drilled by the cable-tool method. Sand and gravel only was encountered at each of the sites. The piezometers were set in place using the pull-back method where the steel casing is withdrawn towards the surface. Piezometers were constructed of PVC (Schedule 40) thermoplastic casing with flush-threaded joints and a 0.9 m (3 foot) length section of slotted PVC 0.5 mm (20 slot) screen connected to a 0.9 m (3 foot) blank tail pipe with closed bottom. A 2.4 m (8 foot) length of screen was utilized in the shallowest piezometer at each site with the screen straddling the water table. Screened portions were sand-packed with Monterey F9 silica sand using a tremie while the steel casing was being withdrawn. About 3 m (10 feet) of

the steel casing was left in the ground to provide a surface seal and protective cap. The annulus between the piezometers and the steel casing was sealed with two thin layers of bentonite and portland cement grout to surface. Each piezometer was developed for approximately 30 minutes with a suction pump until the water was clear. The piezometers were then left undisturbed for two months before water sampling was undertaken.

Prior to sampling in June 1988, water levels were measured in each of the piezometers using the wetted-tape method (U.S. Department of the Interior, 1980). Groundwater samples for laboratory analysis of major chemical parameters, metals and nitrogen isotopes were then taken using a conventional gas-driven suction pump after purging each well for approximately 10 to 20 minutes at rates between 0.3 and 0.6 Lps (5 and 10 USgpm) to remove approximately 380 L (100 U.S. gallons) before sampling. Samples were also taken at the start and end of the pumping for field determinations of temperature, pH, specific conductance and $\text{NO}_3\text{-N}$ concentration for comparative purposes. Samples for dissolved metal determinations were taken following the previously described procedures. Major ion and metal analyses were undertaken at the Environmental Laboratory of the Ministry of Environment in Vancouver. Samples for $\delta^{15}\text{N}$ analysis were field filtered, packed in cooler chests with ice packs and sent by courier within 24 hours to the Agriculture Canada, Research Station in Lethbridge, Alberta for analysis.

Hydraulic head measurements from the piezometers in June, 1988 (Figure 5) indicated a southwesterly groundwater flow direction under a horizontal hydraulic gradient of 0.001. Vertical hydraulic gradients in the range 0.001 to 0.002 were also measured indicating a downward component of flow at Site A and B and an upward component of flow from the two deeper piezometers at Site C. Groundwater recharge conditions at all sites were prevalent to a depth of 11 m (35 feet).

Results of analyses for major chemical parameters, selected dissolved metals and $\delta^{15}\text{N}$ are shown in Tables 1 and 2. Concentrations of a number of parameters were plotted against piezometer intake depths to assess variations with depth. Depth profiles for $(\text{NO}_3+\text{NO}_2)\text{-N}$, Chloride, Specific Conductance, pH, Total Alkalinity, Dissolved Calcium, Sulphate, Dissolved Phosphorous and $\delta^{15}\text{N}$ are shown in Figures 6, 7 and 8. Scatter plots of $(\text{NO}_3+\text{NO}_2)\text{-N}$ versus a number of parameters were also prepared, three of which (Chloride, Specific Conductance and $\delta^{15}\text{N}$) showed significant correlations with $(\text{NO}_3+\text{NO}_2)\text{-N}$. These are shown in Figure 9. Following is a brief discussion of the apparent water quality trends with depth at the sites and observed parameter correlations.

Nitrate-N

As $\text{NO}_2\text{-N}$ concentrations in the area have generally been found not exceeding the detectable limit of 0.005 mg/L, the reported values of $(\text{NO}_3+\text{NO}_2)\text{-N}$ were regarded to be equal to $\text{NO}_3\text{-N}$. $\text{NO}_3\text{-N}$ concentrations (Figure 6) show a decreasing concentration with depth at Sites B and C from 20 to less than 5 mg/L and a slightly increasing trend with depth at Site A (Figure 6). Recharge conditions within the upper 11 m (35 feet) and decreasing nitrate concentrations with depth suggest the downward movement of leachate and nitrate derived from surface sources is actively taking place near Sites B and C. The highest levels of $\text{NO}_3\text{-N}$ at 20.6 mg/L were observed in the 11m (35 foot) deep piezometer at Site C. These trends suggest that the nitrate is derived from surface sources upgradient of Sites B and C but downgradient or to the east of Site A. Concentrations in all piezometers were close to 5 mg/L at a depth of 17 m (55 feet). Decreasing nitrate concentrations with depth also suggest that denitrification processes may also be active. Denitrification with depth has been observed elsewhere in sandy soils (Gillham and Cherry, 1978; Hendry et al, 1983).

Chloride

Chloride concentrations (Figure 6) also exhibit a decreasing trend with depth, from 12 to 4 mg/L at Sites B and C while showing a slightly increasing trend from 2 to 5 mg/L at Site A. These results are similar to the nitrate trends suggesting a surface source possibly derived from animal wastes. A relatively high correlation (Correlation coefficient = 0.82) exhibited between chloride and $\text{NO}_3\text{-N}$ concentrations (Figure 9) suggests a common source. Chloride is a major contaminant often associated with animal wastes (Miller, 1980).

Specific Conductance

Specific conductance, (Figure 6) indicative of the total dissolved mineralization in the groundwater also exhibits a decreasing trend with depth from 280 to 200 $\mu\text{S}/\text{cm}$ at Sites B and C while an increasing trend from less than 100 to 226 $\mu\text{S}/\text{cm}$ occurs at Site A. This appears consistent with the $\text{NO}_3\text{-N}$ and chloride trends indicative of a surface source of pollution situated upgradient of Sites B and C and downgradient or to the east of Site A. A relatively high correlation (Correlation coefficient = 0.64) between specific conductance and $\text{NO}_3\text{-N}$ concentrations suggests a leachate source higher in dissolved mineralization (> 150 mg/L) than ambient groundwaters found at depth (23 m [75 feet]) at Sites B and C.

pH

Laboratory pH values at all of the sites (Figure 7) show an increasing trend with depth from 6 to 8 pH units. Field pH values (Table 1) fall within a similar

range. Field pH values for samples taken nearest the watertable are significantly higher than laboratory pH values for the same samples. This suggests that in situ partial pressures of CO₂ near the water table probably occur under less than 1 atm. conditions. Following equilibrium with CO₂ in the atmosphere, pH of the samples would be expected to decrease. The increasing pH trend with depth may suggest that nitrate is derived from leachate sources more acidic than the ambient groundwaters. Relatively acidic groundwaters in a pH range from 4.2 to 7.5 for example have been associated with contamination from animal waste elsewhere (Ritter and Chirnside, 1984). Alternatively, increasing pH with depth may reflect the dissolution of carbonate minerals (increasing alkalinity and dissolved calcium) as groundwater moves downwards.

Total Alkalinity

Total alkalinity indicative of bicarbonate concentration shows an increasing trend with depth at all of the sites (Figure 7) from less than 10 to 70 mg/L. Low alkalinity groundwaters at shallow depth appears consistent with infiltration of acidic leachate derived from animal wastes.

Dissolved Calcium

Dissolved calcium generally shows an increasing concentration with depth at Sites A and B (Figure 7) from 10 to 30 mg/L but somewhat variable concentration with depth at Site C in the range of 22 to 32 mg/L. Higher concentrations associated with Site C suggest that nitrate sources are also a source of dissolved calcium.

Sulphate

Sulphate concentrations (Figure 8) show an increasing trend with depth at all sites from less than 5 to 22 mg/L suggesting that surface sources of nitrate do not contribute appreciable amounts of sulphate to groundwater. Concentrations close to 20 mg/L probably reflect ambient groundwater values. Site C having the highest sulphate shows some evidence of sulphate reduction occurring below 11 m (35 feet).

Dissolved Phosphorous

Dissolved phosphorous concentrations (Figure 8) show an increasing trend with depth at all sites from 0.003 to 0.014 mg/L. The observed trend may be related to the increasing pH which increases the solubility of phosphorous likely present in the sediments. This suggests that the sources of $\text{NO}_3\text{-N}$ are not significant sources of phosphorous or that adsorption processes within the unsaturated zone significantly attenuate phosphorous movement.

$\delta^{15}\text{N}$

Nitrogen isotope results are expressed as:

$$\delta^{15}\text{N} \text{ (ppt)} = \frac{(^{15}\text{N}/^{14}\text{N})_{\text{sample}} - (^{15}\text{N}/^{14}\text{N})_{\text{standard}}}{(^{15}\text{N}/^{14}\text{N})_{\text{standard}}} \times 1000$$

where atmospheric nitrogen is used as the standard (Kreitler, 1975). Analysis

results were obtained as ‰¹⁵N and converted to δ¹⁵N using ¹⁴N standard = 99.632‰ and ¹⁵N standard = 0.368‰ as reported by Kreitler (1975). Positive values of δ¹⁵N indicate samples enriched in ¹⁵N relative to ¹⁴N while negative values of δ¹⁵N indicate samples enriched in ¹⁴N.

δ¹⁵N values (Figure 8) show an increasing trend with depth at all of the sites in the range of +5.5 to +23.7 ppt. δ¹⁵N values > +9.0 in groundwaters elsewhere have been associated with animal waste leachates (Gormly and Spalding, 1979). This increasing trend with depth is in opposition to the decreasing NO₃-N trend. A relatively high inverse correlation (Correlation Coefficient = -0.81) exists between NO₃-N and δ¹⁵N (Figure 9). This inverse relationship suggests that isotopic fractionation caused by denitrification is most likely an active process working at depth. Similar results in highly permeable, coarse-textured deposits have been observed elsewhere (Gormly and Spalding, 1979). This process likely involves the reduction of nitrates to ammonia, to nitrogen gas or to nitrous oxide by microbial activity. There is, moreover, some evidence of nitrogen reported as ammonia up to 0.034 mg/L occurring in some of the deeper piezometers (Table 1) while near detectable values (0.005 mg/L) only occur in the water-table piezometers. Denitrification by anaerobic bacteria for example would moreover release nitrogen gas enriched in the lighter isotope ¹⁴N leaving nitrate in solution enriched in ¹⁵N (Wellman et al, 1968). Gormly and Spalding (1979) report that fresh animal wastes are not initially enriched in ¹⁵N, but fractionation of nitrogen isotopes appears to commence soon after the wastes are deposited on the ground. Such enrichment results from the volatilization of NH₃ produced by the decomposition of urea, a major component of animal wastes (Kreitler, 1975). This would result in residual aqueous ammonium in leachate being enriched in ¹⁵N creating a source of ¹⁵N for potential nitrification to nitrate.

$\delta^{15}\text{N}$ values showing slight enrichment in ^{15}N near the water table may also be indicative of natural soil nitrogen sources (Kreitler, 1975). Lowering of the water table by pumping and drainage schemes moreover, could be a contributing factor to nitrate formation in the region. Lowered water levels for example would promote ammonification of natural organic nitrogen sources, nitrification of ammonia and formation of nitrates which could be mobilized to the water table by infiltrating waters or a seasonal rise of the water table.

Transport and transformation of nitrogen from animal wastes at the land surface to nitrate in groundwater involves a number of biological, chemical and physical processes including hydrolysis, ammonification, oxidation, nitrification and leaching by percolating waters. Nitrogen in animal wastes principally occurs as organically-bound nitrogen (amino acids, polypeptides and proteins) in feces and as urea (NH_2CONH_2) in urine. Under dry conditions nitrogen in urea can be transformed biologically to ammonium NH_4^+ through hydrolysis of the enzyme urease (Robertson et al, 1974). NH_4^+ can also be produced through mineralization of organic nitrogen (ammonification). In the soil zone NH_4^+ is converted to NO_3^- by oxidation involving nitrifying bacteria (nitrification). Ammonification and nitrification are processes that usually occur above the water table where organic matter and oxygen are abundant (Freeze and Cherry, 1979). Once formed, NO_3^- can be readily leached by percolating waters and transported significant distances in groundwaters under strongly oxidizing conditions.

Other Metals

Analyses of a number of additional dissolved metals including those listed in Table 2 and including cadmium, cobalt, chromium, copper, nickel, lead, and

vanadium showed no particularly anomalous values or discernable trends with depth. Ritter and Chirnside (1984) report that some metals such as copper may be a component of poultry feed. Reported concentrations of dissolved cadmium, chromium, copper and vanadium were all less than the detectable limit of 0.01 mg/L. Concentrations of dissolved cobalt and lead were all less than the detectable limit of 0.1 mg/L and all nickel values were below 0.05 mg/L. Trace quantities of boron, barium, zinc and molybdenum (Table 2) were reported in some of the piezometers, but the significance of these concentrations is unknown. Concentrations of dissolved iron in the range 0.01 to 0.02 mg/L and dissolved manganese in the range 0.01 to 0.14 do not suggest particularly strong reducing conditions at depth. Low sodium concentrations in the range of 2.7 to 6.9 mg/L, low potassium in the range of 0.6 to 5.2 mg/L and low magnesium concentrations in the range of 4.6 to 6.1 mg/L do not show any discernable trends.

Overall Groundwater Quality

In terms of overall groundwater quality based on equivalents per million as a percentage, groundwaters at Site A can be classified as a calcium-bicarbonate type, while groundwaters at Sites B and C contain approximately equal portions of bicarbonate, chloride, nitrate and sulphate anions. Groundwaters are generally low in overall mineralization with TDS values in the range of 76 to 270 mg/L. $\text{NO}_3\text{-N}$ concentrations up to 20.6 mg/L or 92 mg/L as NO_3 can contribute a significant portion (one third) of the dissolved minerals.

SUMMARY AND CONCLUSIONS

A significant portion of the glacial fluvial sand and gravel deposits

comprising the Abbotsford Aquifer is contaminated by nitrates exceeding the provincial drinking water quality standard of 10 mg/L as nitrate-nitrogen. Water quality data obtained from long-term monitoring of a number of sites over a 15-year period indicate that nitrate concentrations vary seasonally. Highest nitrate concentrations generally occur during the fall and winter months when nitrate is leached to the water table during recharge from infiltrating precipitation and rising water table conditions. An increasing trend in nitrate concentration is generally evident in groundwaters within the eastern discharge area where nitrate-nitrogen levels have risen at one site for example from 1.2 mg/L in 1974 to 8.3 mg/L in 1984. Relatively high levels of ammonia-N in the range of 1.3 to 4.7 mg/L have been found in unpurged observation wells indicative of microbial denitrification of groundwater in the immediate vicinity of the well. Adequate purging of monitor wells and determination of all major dissolved nitrogen species is necessary for interpreting the significance of nitrate concentrations in groundwaters. Site specific determinations of water quality with depth at a control site instrumented with 12 standpipe piezometers indicate that nitrate concentrations decrease with depth. Nitrates up to 21 mg/L as nitrogen can contribute up to a third of the total dissolved mineralization at some locations. Groundwater high in nitrates correlates significantly with increased concentrations of chloride and specific conductance suggesting an organic source of contaminants such as animal wastes. Absence of tritium in groundwater from domestic wells as shallow as 10 m in depth suggests a relatively long history of nitrate contamination probably originating after World War II. $\delta^{15}\text{N}$ isotope determinations indicate enrichment of ^{15}N with values of $\delta^{15}\text{N}$ in the range of +5.5 to +23.7 ppt. indicative of nitrogen derived from organic sources such as soil humus and animal wastes. An inverse correlation between nitrate and $\delta^{15}\text{N}$ indicates that denitrification processes are active at depth reducing nitrate

concentrations and enriching groundwaters in ^{15}N . These results and observations of land use practices indicate that poultry manure wastes which are inappropriately stored, disposed of and applied to the land are a principal source of nitrate contamination of groundwaters in the area.

Additional studies which would provide further evidence of nitrogen species transformation, nitrate sources and movement might include nitrogen analyses of soil samples, nitrogen analyses of poultry litter leachates, water sampling in the unsaturated zone using lysimeters, microbiological sampling and analyses in the unsaturated zone and in groundwater, measurements of Eh in groundwater and more comprehensive isotope analyses. More frequent monitoring (e.g. weekly) of nitrate concentrations during the fall and winter recharge period would provide more definitive information on temporal variations.

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TABLE 1

Results of Major Chemical Parameter and N Isotope Analyses at Piezometer Sites

Site/Depth		Lab. Field Spec.										
Feet	NO ₃ +NO ₂ (N)	NH ₃ (N)	Org.N	Phos.Diss.	Total Alk.	pH	pH	Cond.	TDS	Cl	SO ₄	δN ¹⁵
A-25	4.55	<0.005	<0.01	.004	17.8	6.5	7.2	88	94	3.5	2.3	14.21
A-35	3.5	<0.005	<0.01	.005	24.3	6.6	6.8	88	76	2.3	2.4	15.27
A-55	5.2	.006	<0.01	.007	53.3	8.1	8.2	165	142	2.6	3.5	12.54
A-75	5.5	.014	<0.01	.008	65.6	8.3	8.6	226	166	5.4	16.6	19.11
B-20	16.8	.008	<0.01	.003	8.7	6.9	7.7	220	228	10.3	9	5.46
B-35	16.6	.005	<0.01	.003	14.1	7.3	7.2	217	240	8.4	8.9	7.28
B-55	6.55	.034	.03	.008	52.9	7.6	7.8	196	160	4.5	9	13.29
B-75	2.3	<0.005	.04	.01	71.1	8.2	8.3	212	148	4.1	16.9	N/A
C-25	17.1	.005	.03	<0.003	9.8	6	7.6	234	202	10.2	14.7	8.35
C-35	20.6	.02	.03	<0.003	13	6.3	6.9	283	270	12.1	16.3	7.31
C-55	4.9	.02	<0.02	.007	50.8	7.3	7.7	184	140	4.6	11.3	23.66
C-75	.58	.008	.06	.014	64.8	8.1	8.3	210	146	7.9	21.5	N/A
Average:	8.68	.0133	.038	.007	37.2	7.3	7.7	194	168	6.33	11	12.65
Maximum	20.6	.034	.06	.014	71.1	8.3	8.6	283	270	12.1	21.5	23.66
Minimum:	.58	<0.005	<0.01	<0.003	8.7	6	6.8	88	94	2.3	2.3	5.46

All analyses in mg/L except pH (relative units), Specific Conductance (μS/cm), and δN¹⁵ (ppt).

TABLE 2

Results of Selected Dissolved Metals Analyses at Piezometer Sites

Site/Depth	Na	K	Ca	Mg	B	Ba	Fe	Mn	Zn	Mb
A-25	2.7	.6	10.7	1.89	<0.01	<0.01	.02	.02	<0.01	<0.01
A-35	2.9	.6	10.8	1.95	<0.01	<0.01	<0.01	.03	<0.01	<0.01
A-55	3.4	1	20.3	4.94	<0.01	<0.01	<0.01	.03	<0.01	.01
A-75	4.1	1.9	31.5	5.78	<0.01	<0.01	<0.01	.03	<0.01	.01
B-20	4.1	5.2	22.7	4.26	<0.01	.07	<0.01	.01	.01	<0.01
B-35	4.7	1.8	24.1	4.69	<0.01	<0.01	<0.01	.02	<0.01	<0.01
B-55	4.4	1.4	22.1	5.53	<0.01	<0.01	<0.01	.01	<0.01	<0.01
B-75	4.7	1.8	27.7	4.89	<0.01	.01	<0.01	.01	<0.01	<0.01
C-25	5.5	1	24.9	5.22	.01	.01	.01	.04	<0.01	<0.01
C-35	6.1	1.2	31.7	6.07	.01	<0.01	<0.01	.14	<0.01	<0.01
C-55	4.7	1.1	21.6	4.82	<0.01	<0.01	.01	.01	<0.01	<0.01
C-75	6.9	1.6	26.2	4.82	<0.01	<0.01	<0.01	.03	<0.01	<0.01
Average:	4.5	1.6	22.9	4.57	<.01	.03	.01	.032	<0.01	<0.01
Maximum:	6.9	5.2	31.7	6.07	.01	.07	.02	.14	.01	.01
Minimum:	2.7	.6	10.7	1.89	<.01	<0.01	<.01	.01	<0.01	<0.01

All analyses reported in mg/L.

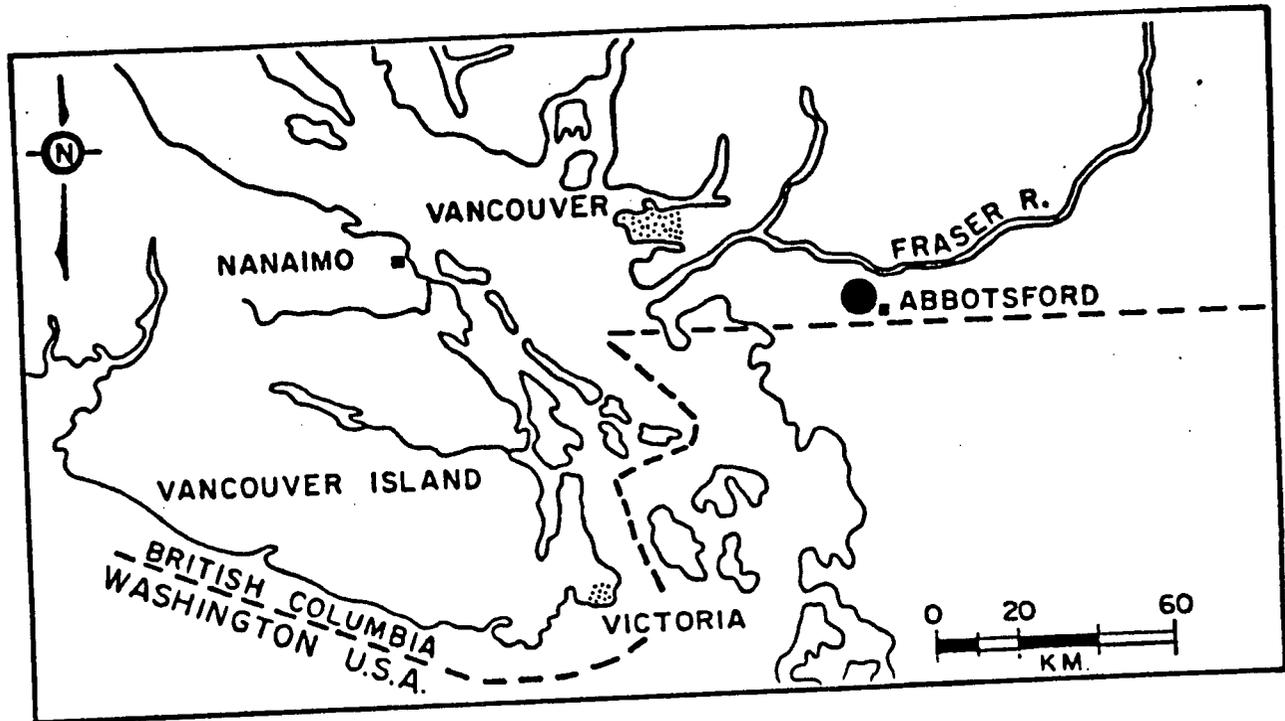


Figure 1. Location map.

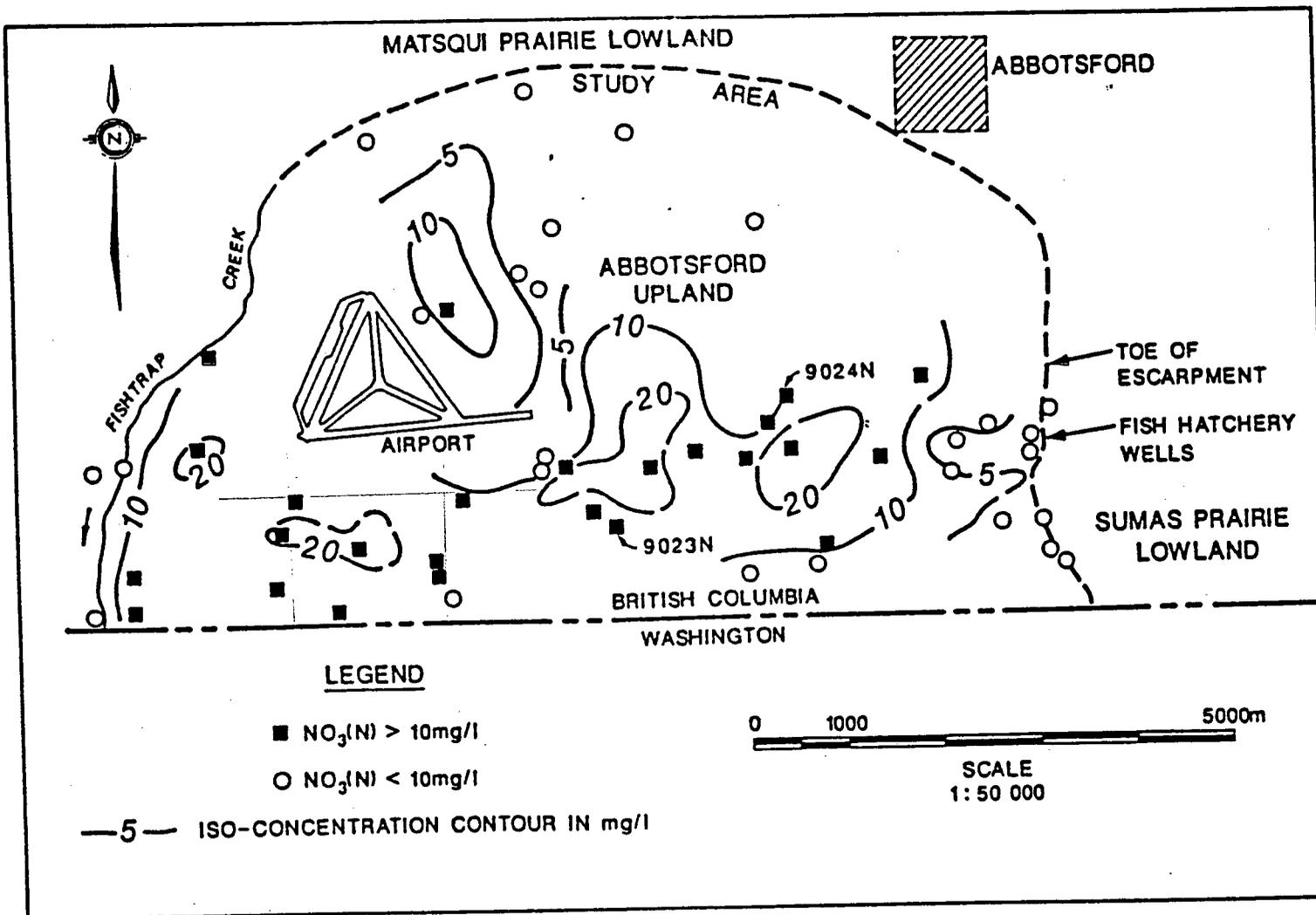


Figure 2. Occurrence of nitrate-nitrogen in domestic wells, Abbotsford Upland (after Kwong, 1986a)

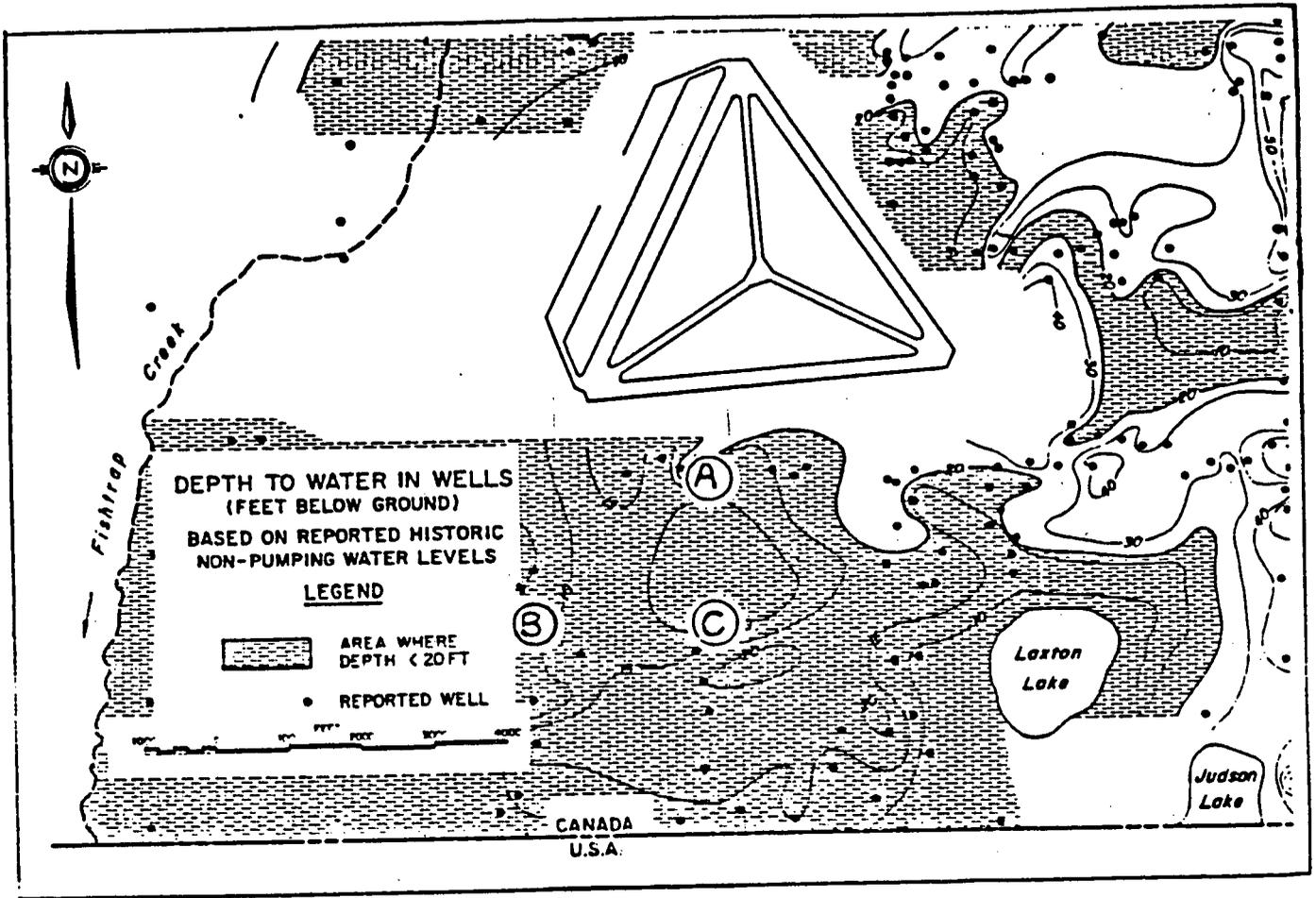


Figure 3. Depth to water and location of piezometer sites A, B and C south of Abbotsford Airport (after Kohut, 1987).

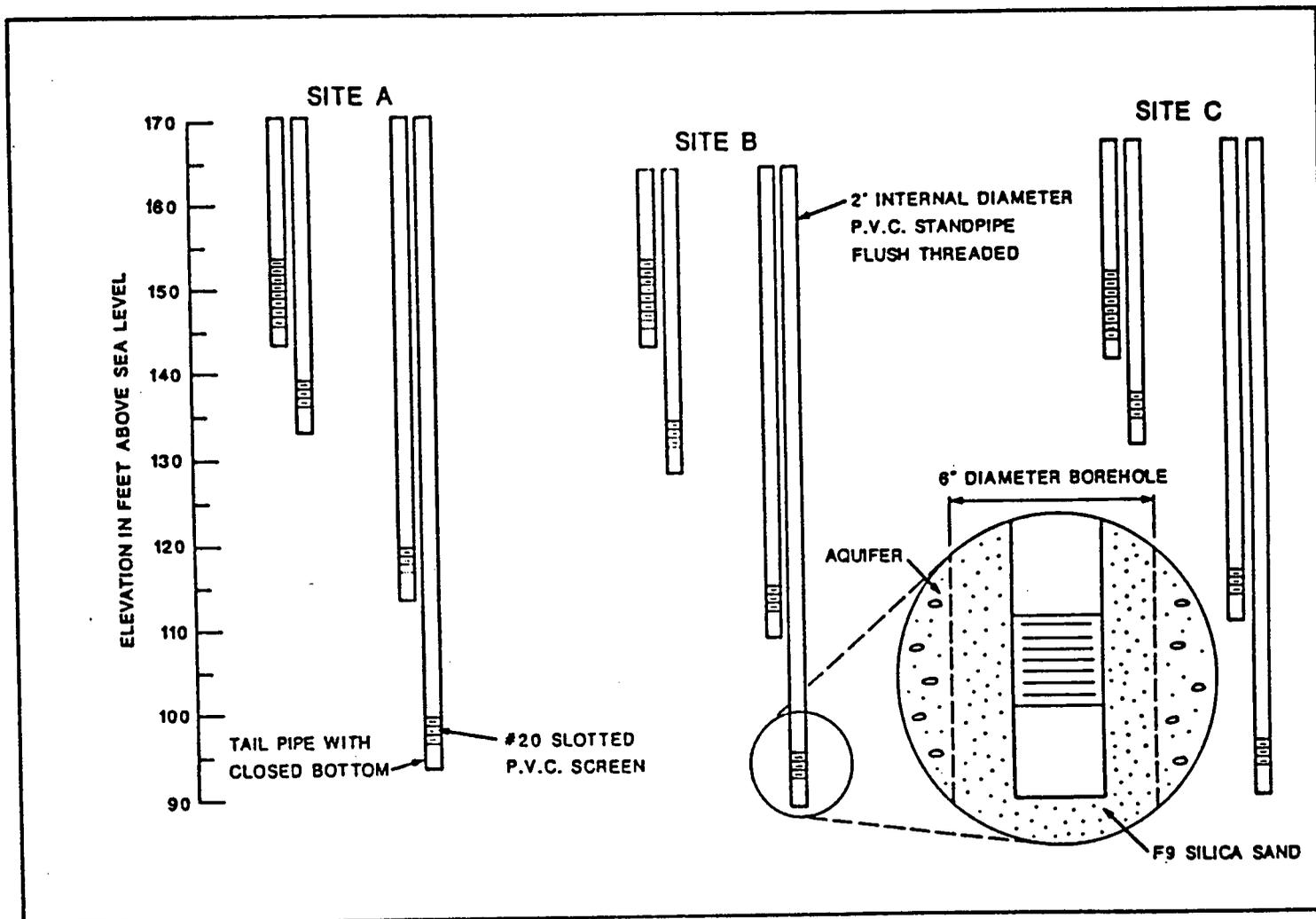


Figure 4. Piezometer settings and construction details.

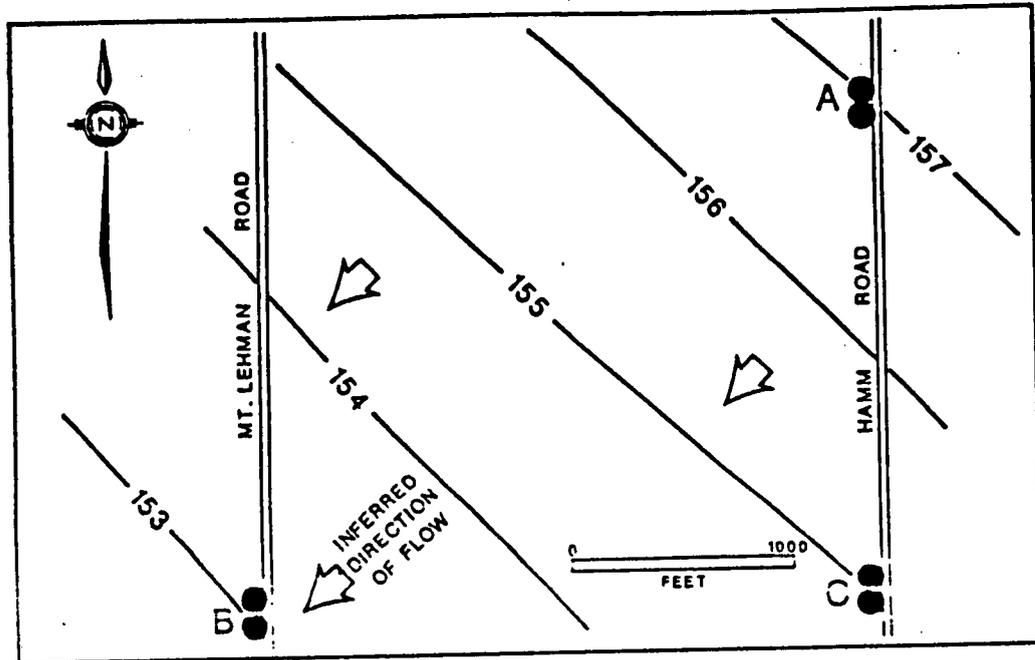


Figure 5. Water table elevation contours, in feet above sea level and inferred direction of flow between piezometer sites, June 1988.

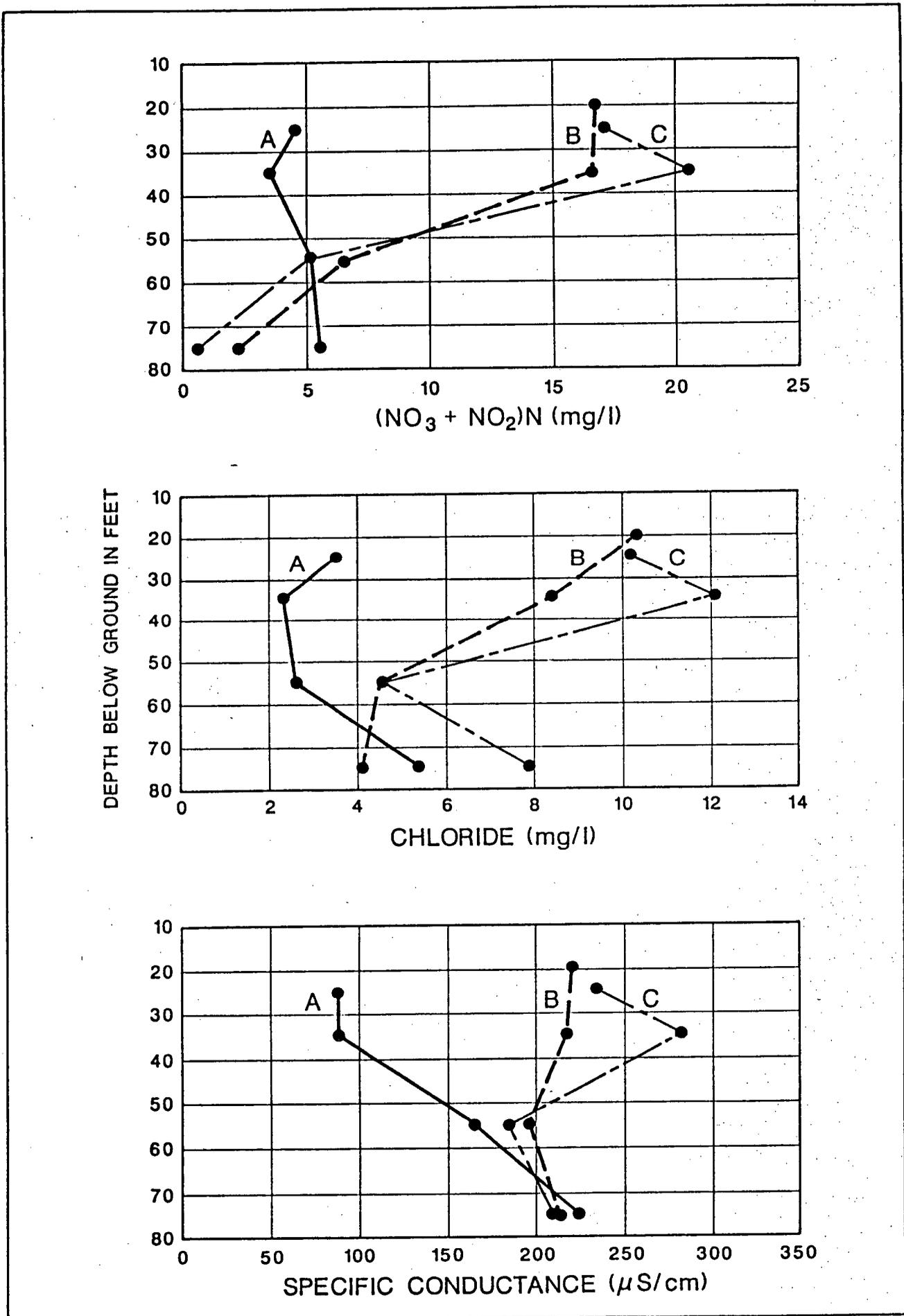


Figure 6. Variations of (NO₃ + NO₂)N, Chloride and specific conductance with depth.

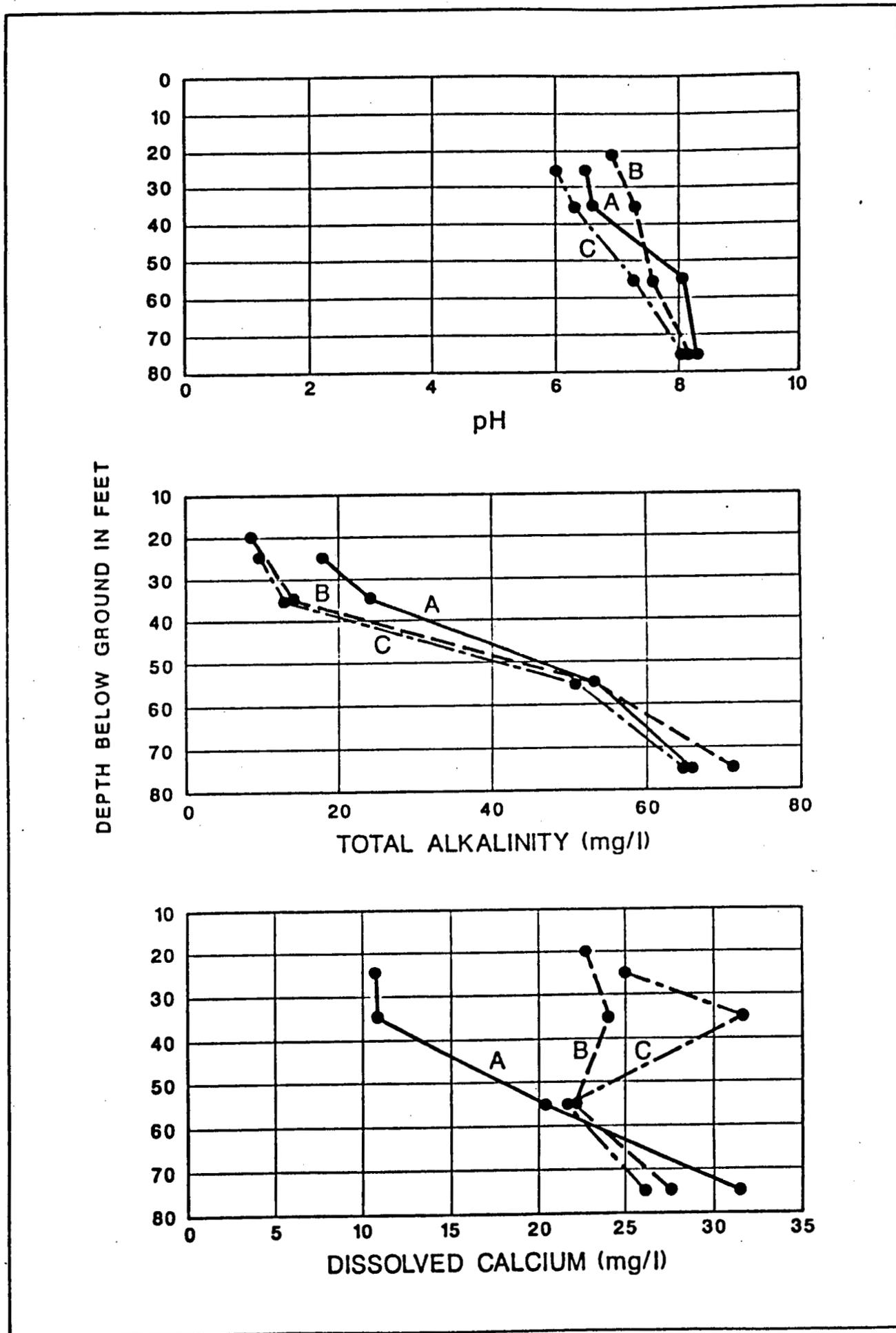


Figure 7. Variations of pH, total alkalinity and dissolved calcium with depth.

DEPTH BELOW GROUND IN FEET

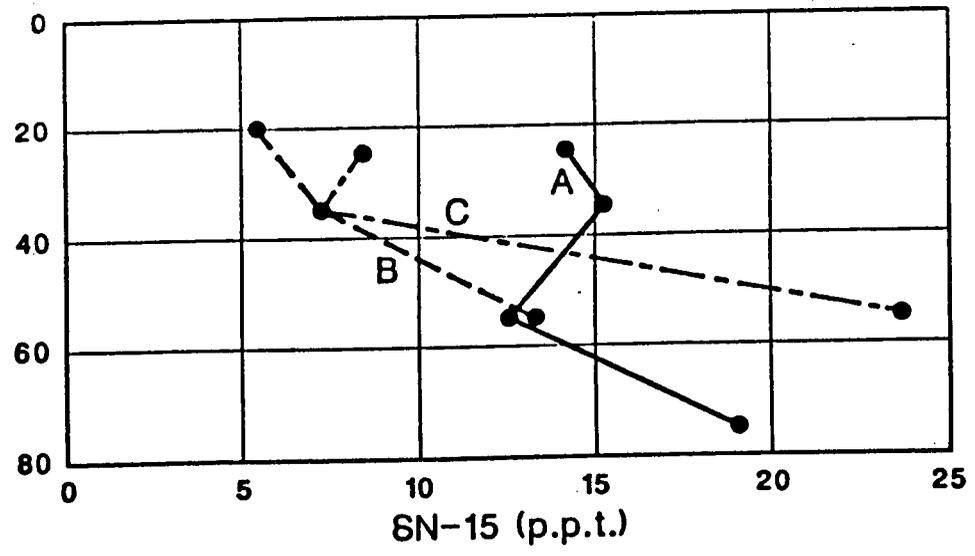
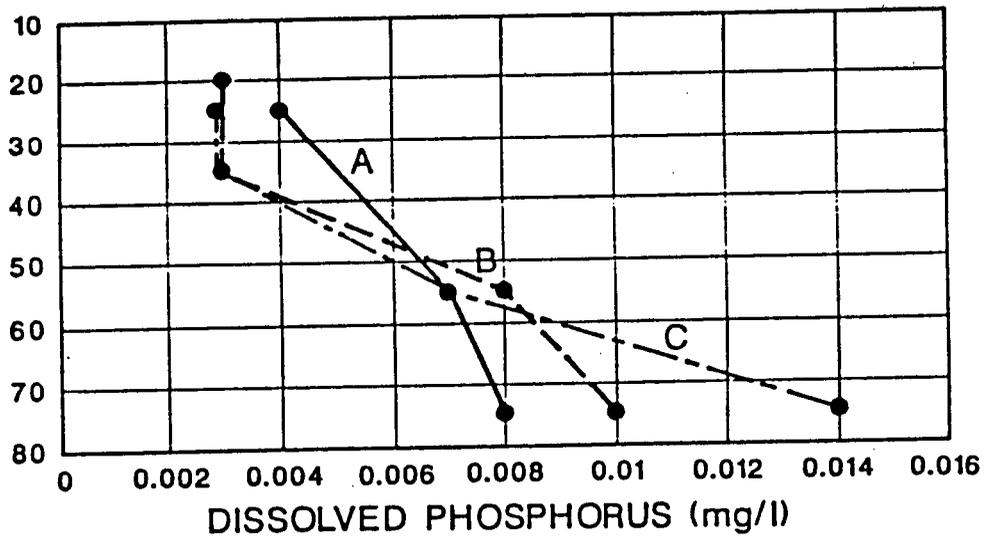
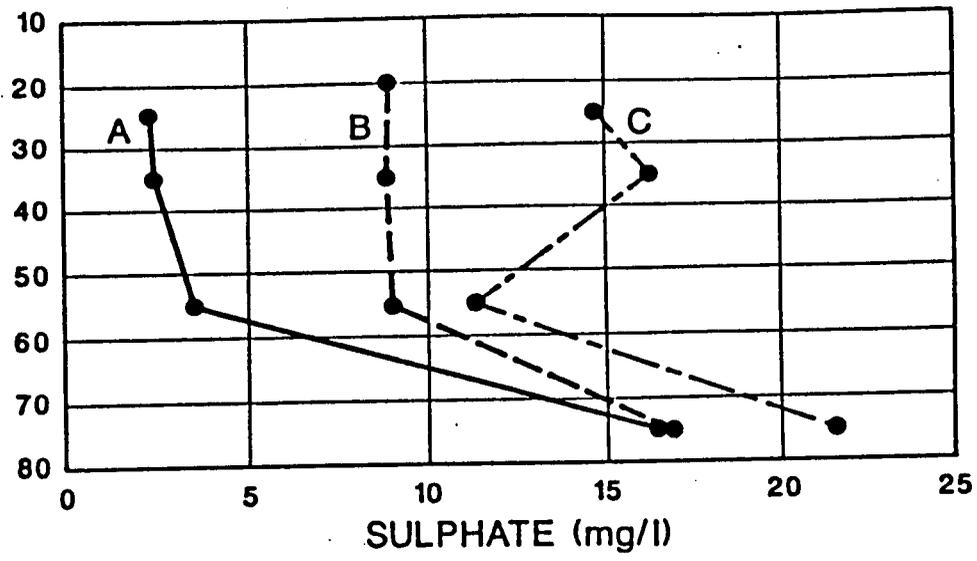


Figure 8. Variations of sulphate, dissolved phosphorus and δN^{15} with depth.

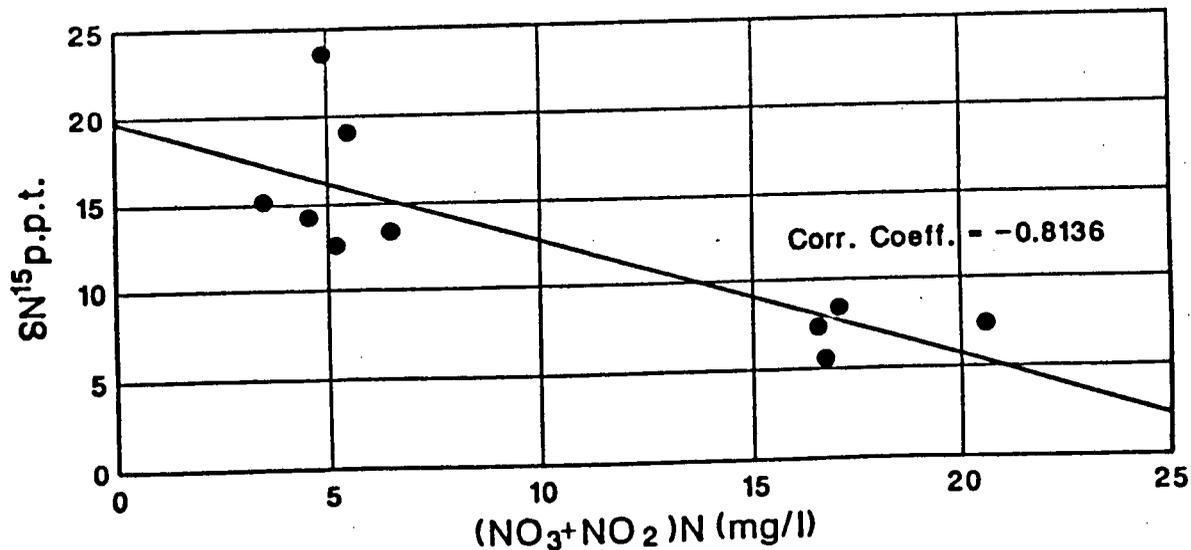
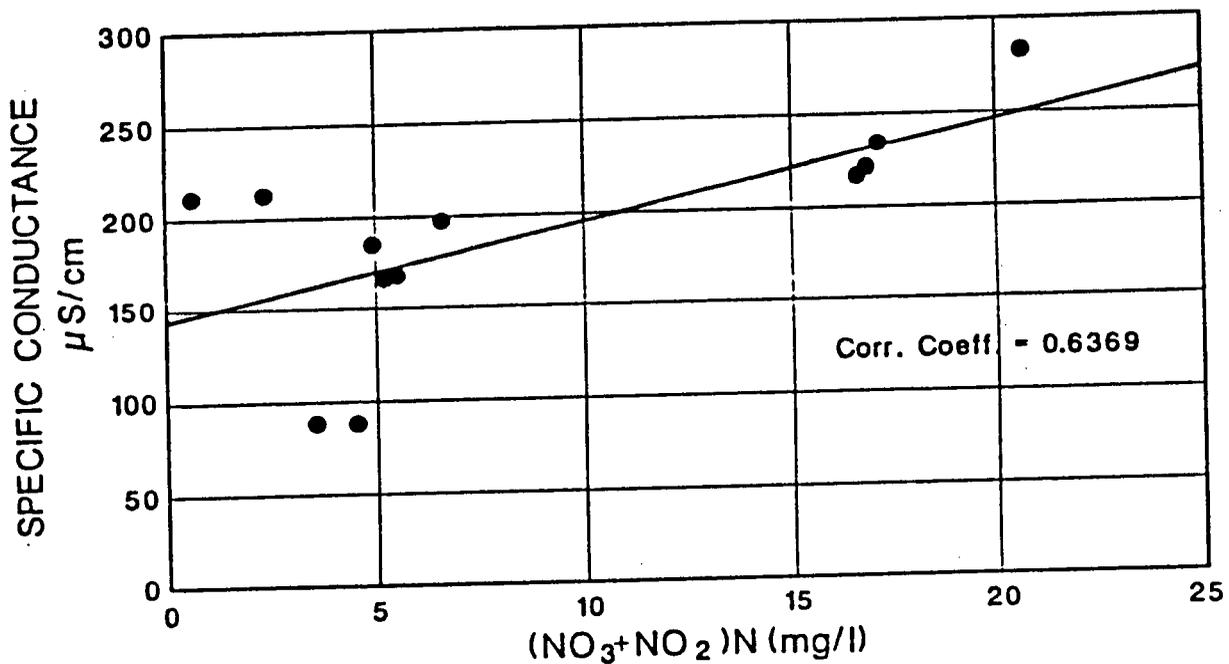
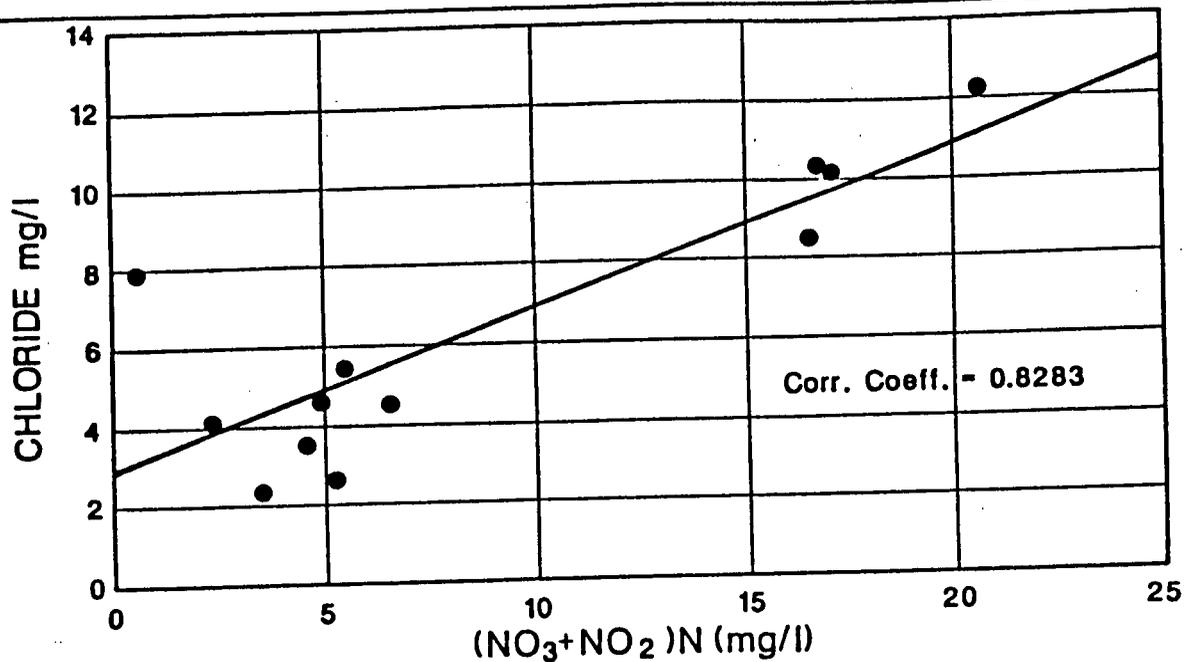


Figure 9. Scatter diagrams of chloride, specific conductance and δN¹⁵, versus (NO₃ + NO₂)N.