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Hydrochemical Variations Through Saturated Overburden Deposits Beneath an Irrigated Site in South Central Alberta

by

Deborah Pike-Glover

A THESIS
SUBMITTED TO THE FACULTY OF GRADUATE STUDIES AND RESEARCH
IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE DEGREE
OF MASTER OF SCIENCE

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The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research, for acceptance, a thesis entitled Hydrochemical Variations Through Saturated Overburden Deposits Beneath an Irrigated Site in South Central Alberta submitted by Deborah Pike-Glover in partial fulfilment of the requirements for the degree of Master of Science.

D. L. Brown

Supervisor

David W. Schwartz

Arleigh H. Laycock

Date March 1st 1982

Abstract

This study was undertaken at an irrigated si north of Taber, Alberta. The study site compr plain, underlain by laterally and vertically Laurentide glacial, fluvioglacial, fluvial deposits which disconformably overlie Creta y bedrock at depths of forty to fifty meters d sequence of deposits from the surface to b ace sand, fractured oxidized till, buried out ssive non-oxidized till, Saskatchewan sands and avels and bedrock.

The extensive network of piezometers and water table monitoring wells across the study site shows that the water table is generally shallow, being between 2 and 5 meters beneath ground surface although locally it reaches depths of 13 to 14 meters. Groundwater flow is dominantly vertically downward. Lateral flow occurs in places through sand and gravel lenses in the tills and in the surface sands and Saskatchewan sands and gravels.

5:1 aqueous extracts obtained from bulk samples of surface sand and oxidized till show that the till contains much higher concentrations of water soluble salts than the surface sands. An extract from a buried sand sample is generally comparible with till extracts.

Results of analyses for three groundwater sample sets are presented. Field and laboratory analysis results are processed using WATEQF, a computer program which gives the

equlibrium speciation for each sample under the temperature and pH conditions existing in the field.

Groundwater samples from piezometer nests 1811 and 1818 have much lower concentrations of major ion species than samples from elsewhere in the study site. No reasons for this can be identified.

Shallow groundwater samples from just beneath the water table often have very high ionic concentrations (e.g. TDS=11,500mg/L). This is a result of a combination of processes in the unsaturated zone including carbon dioxide production and dissolution, pyrite oxidation, dissolution of carbonates and sulphates, and water losses by evaporation and evapotranspiration.

Concentrations of major ion species in the groundwater vary through the profile. Sodium and sulphate are usually the dominant ions. No definite pattern exists of changes in groundwater chemistry through the different overburden units. The chemistry changes observed represent the end result of a combination of modifying processes. Since the relative magnitude of processes and effects are unknown, it is impossible to define the processes which lead to the observed complex response in groundwater chemistry.

Cation exchange occurs in the smectite rich non-oxidized till but is restricted by rapid fracture flow velocities in the oxidized till. Denitrification in the tills acts as an adequate decontamination mechanism to prevent high nitrate-nitrogen concentrations from agricultural or geologic sources from reaching the Oldman River. High nitrate-nitrogen concentrations in the buried sand deposits may pose future problems. Further research would be necessary in order to investigate this aspect of the groundwater chemistry.

Preface

The site at which this study was undertaken was chosen by Alberta Agriculture in 1978 to provide insight into a flow system which would contrast with a study Alberta Agriculture were undertaking at the time in the Bow River Irrigation District, (BRID). The investigation in the BRID also considered a flow system and the quality of return flow water in overburden deposits, although at that site lateral flow formed the major component of the groundwater flow system and the return flow entered the Bow River. In contrast, in this study flow was dominantly vertically downward. Also, the BRID site had been irrigated over a much longer time span than the site investigated in this thesis.

Groundwater monitoring instrumentation, consisting of a network of piezometers and water table wells was installed at the study site which is investigated here, between June 1978 and the summer of 1980 when this study was undertaken. During that period groundwater elevations in water table observation wells and piezometers were monitored monthly by the Drainage Branch of Alberta Agriculture. This enabled construction of a comprehensive model of the groundwater flow system beneath the site.

Pollution of rivers by saline return flows from irrigation practices is a well documented problem in southern Alberta. The study site considered here may not be indicative of the situation throughout southern Alberta, however it is hoped that this study will add to the knowledge of processes contributing to southern Alberta's salinity problems.

Acknowledgements

This study could not have been undertaken without the financial support of the Drainage Branch of Alberta Agriculture. The author is extremely grateful for this funding as well as the encouragement and advice of the Drainage Branch staff. Specail thanks are extended to Mr. M.J. (Jim) Hendry for his supervision and encouragement throughout the study. Thanks are also extended to Alberta Environment and the staff of the Alberta Environment Water Quality Laboratory in Lethbridge for their cooperation in analysis of water samples.

Dr. M.C. (Charlie) Brown's advice and supervision at the University of Alberta were greatly appreciated as were the comments of Dr. A. Laycock and Dr. F. Schwartz. Dr. Schwartz's comments on atmospheric sulphate input and Dr. Laycock's extensive knowledge of water balance in southern Alberta added to the understanding of the system investigated in this study.

Dr. J. Honsaker's advice on production of this thesis on the computer are gratefully acknowledged.

Finally, but by no means least, special thanks go to my husband for his encouragement and patience, and to my parents for their financial support which enabled me to persue my university studies and to come to Canada.

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1. Introduction

This thesis is half of a two part study undertaken at an irrigated site in south central Alberta during the Summer and Fall of 1980. G. Burnett of the University of British Columbia performed the other part of the study which he presented as his Masters Thesis (Burnett, 1981). The two studies are being compiled into a single report by the Drainage Branch of Alberta Agriculture.

The objectives of this study were threefold;

1. To determine the quality of groundwater and the variations in quality beneath the study site

2. To develop a hydrochemical evolution model which could account for the observed changes in groundwater chemical quality as it passes through the overburden deposits,

3. To use groundwater flow data from the Taber Irrigation District (TID) Return Flow Study Report (Alberta Agriculture, 1981) in conjunction with the groundwater quality data collected here to assess the present and future effects of irrigation return flow on the quality of the Oldman River.

The study is confined to the saturated zone within the overburden deposits. Extraction of water samples from the unsaturated zone is difficult and was not attempted. Also, only limited hydraulic connection exists between the overburden deposits and the bedrock so that only flow within the overburden deposits is considered here.

2. Physiography, Overburden Deposits, and Hydrogeology

The study site is located 15km north of Taber and 60km east of Lethbridge (figure 1), and encompasses approximately 34km^2, it is bounded on the west and north sides by the Oldman River. Highway 36 forms the east boundary to the site while the south boundary is defined by the line separating sections 18 and 19, 17 and 20, and 16 and 21, (Tp.11, Rg.16).

This south-central region of Alberta experiences a semi-arid climate with approximately 360mm mean annual precipitation (Hydrological Atlas of Canada) with most of this precipitation falling between April and September. Snow often covers the area during the winter and redistribution by wind leads to uneven snow cover, variable freezing depths and spatially non-uniform recharge on thawing. Intermittent rapid thawing occurs during the winter as a result of Chinook activity. At these times evaporation rates are high and runoff is rapid so that the resulting recharge to the water table can sometimes be insignificant.

Temperatures in the area are often extreme with an annual range from maximum to minimum temperature in the order of 75 degrees celsius. High summer temperatures and the windiness of the region result in high evapotranspiration rates. This, coupled with the aridity of the area, has necessitated the development of extensive irrigation networks in order to increase crop yields over those obtained by former dryland farming.

2.1 Soils

The soils of the study area are classified as Orthic Brown Chernozems being predominantly the sandy loams and loamy sands of the Cavendish group (Bowser et al., 1963). These soils are developed from the underlying glacial and fluvioglacial deposits and are weakly to moderately calcareous in the lower horizons. They allow rapid infiltration and have medium to high permeabilities accompanied by a low water holding capacity due to their relatively low organic content and high percentage of sands, although they are fairly well suited to crop growth when irrigated. Drainage is a potential problem since till underlies the Cavendish Group soils at an average depth of 1.5m, and the upper surface of the till deposit forms a semi-permeable boundary to vertical water movement because of the till's relatively low hydraulic conductivity. This sometimes leads to water-logging and lateral movement of water downslope at or above the contact between the soil and the underlying till (Bowser et al., 1963), hence, good irrigation management is necessary to avoid drainage problems.

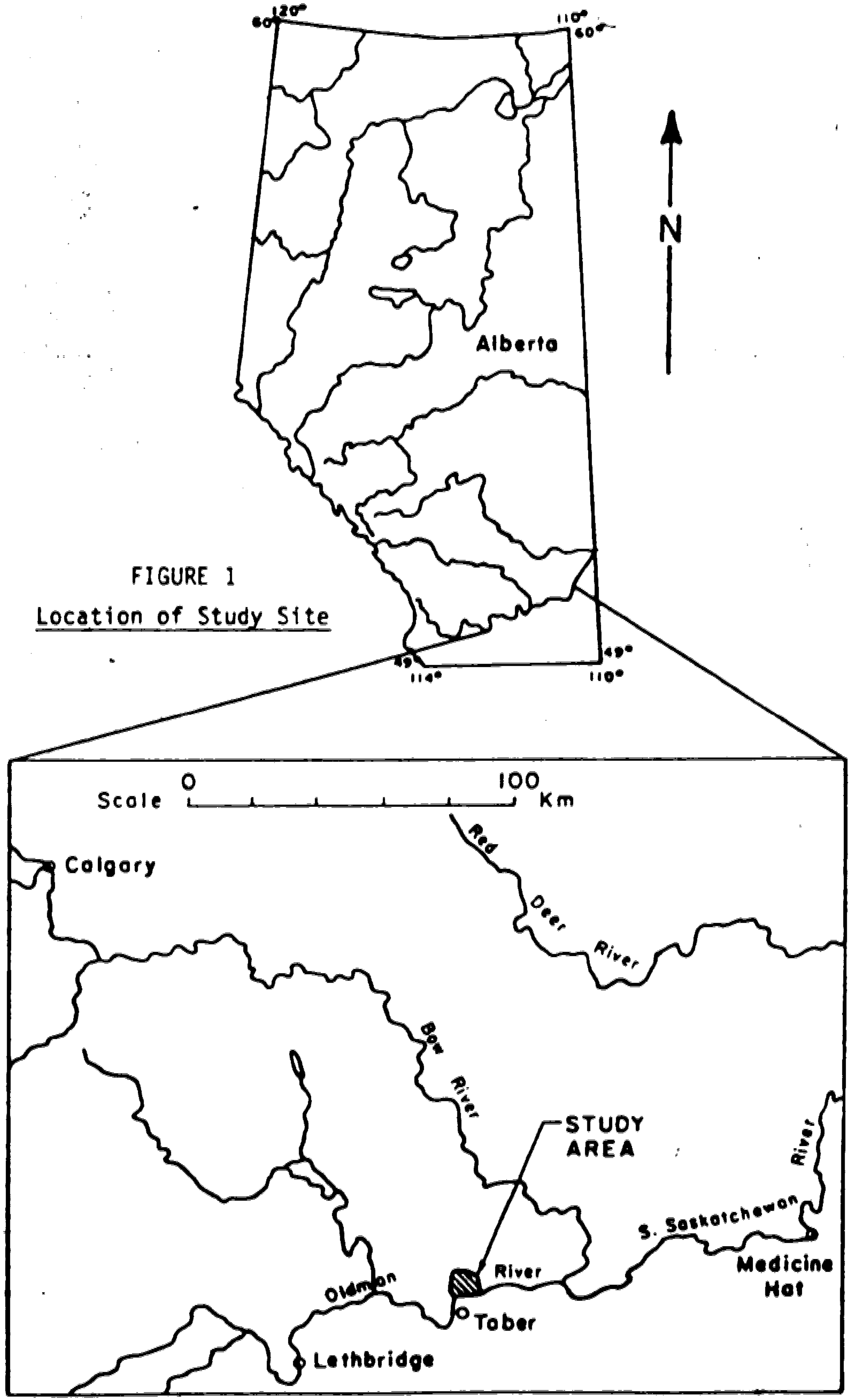

FIGURE 1

Location of Study Site

2.2 Topography

The topography of the field area is hummocky due to the underlying glacially derived deposits and wind blown sands. A topographic contour map of the area is shown in figure 2. Some of the depressions contain standing water since the water table is generally shallow and at times intersects the ground surface. The major relief in the area is governed by the Oldman River valley which has dissected the overburden deposits to a depth of approximately 40m, following the probable course of the preglacial Oldman valley as suggested by Farvolden (1963). The outside of the bend is a steep bank cut by the river, however the inside bank has a more moderate slope with some terrace development and lag gravel deposits which include large boulders. Steep sided coulees have been developed by intermittent stream activity along both banks of the Oldman river.

2.3 Land Use

Land use in the field area is illustrated in figure 3 and falls into two categories:

1. The irrigated areas which are used intensively for agriculture and comprise approximately 65% of the total area. Pivot and sideroll sprinklers are used to produce wheat, barley, corn, potatoes, carrots, onions, and sugar beets.

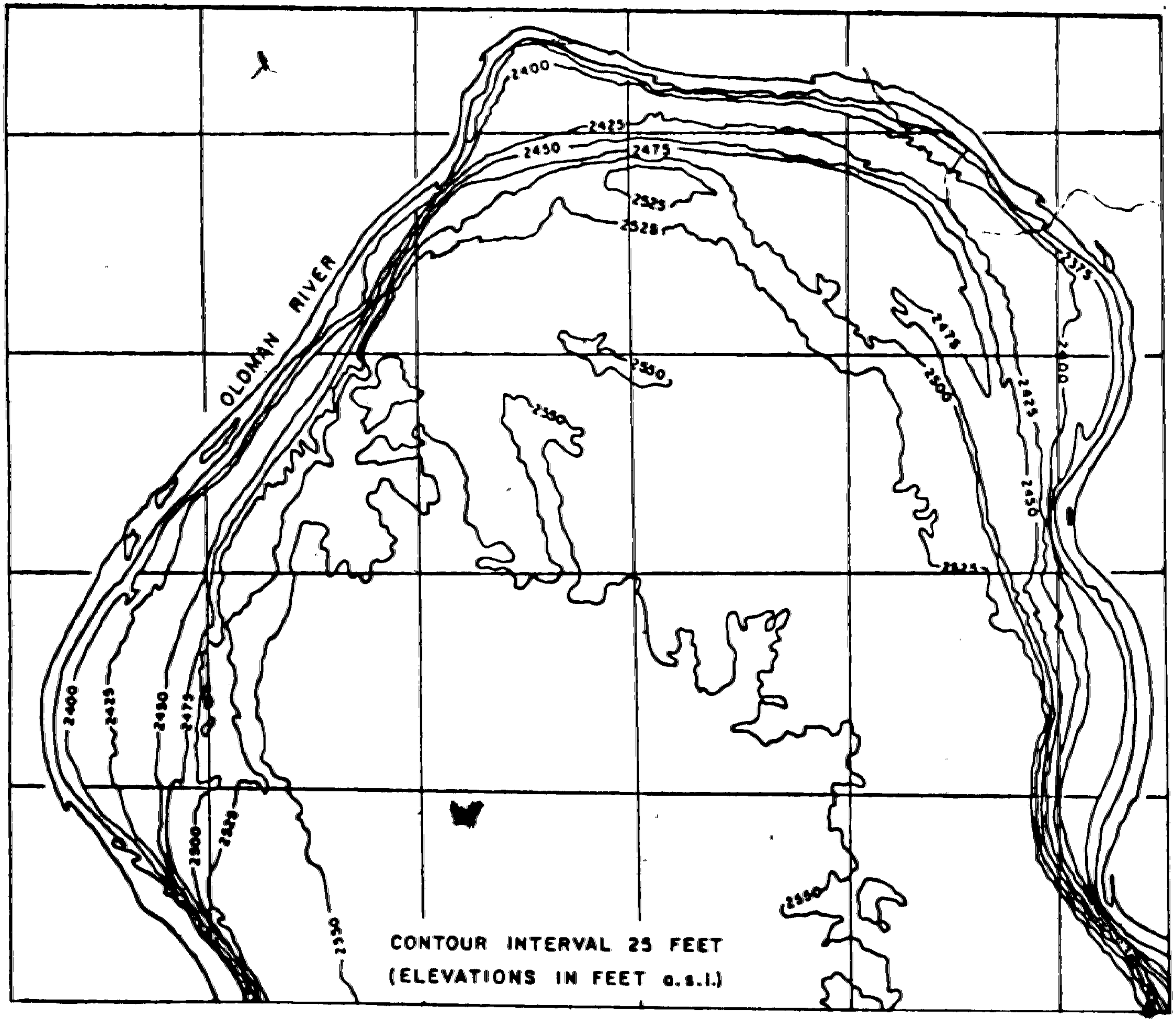

FIGURE 2

Topographic Contour Map

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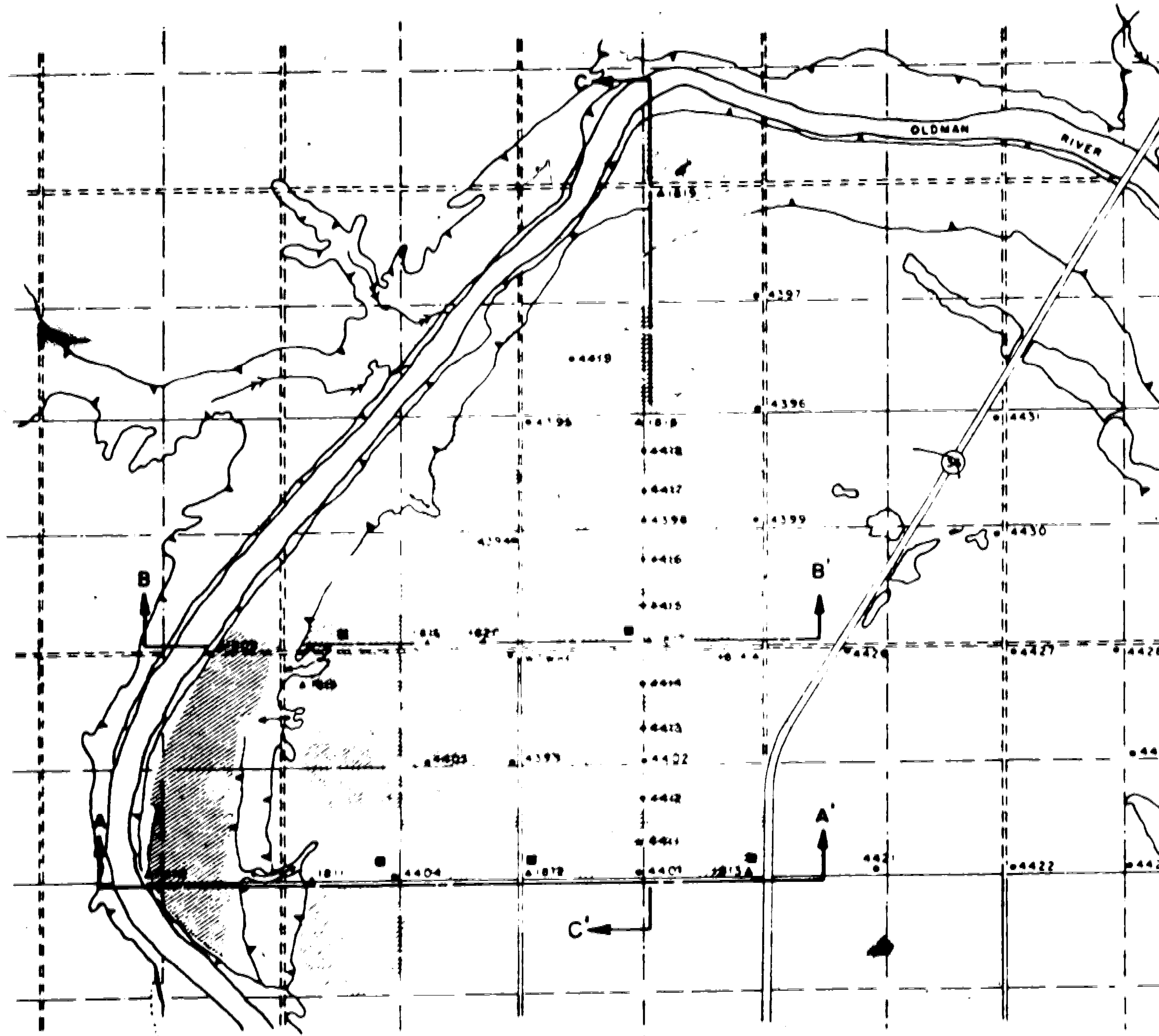

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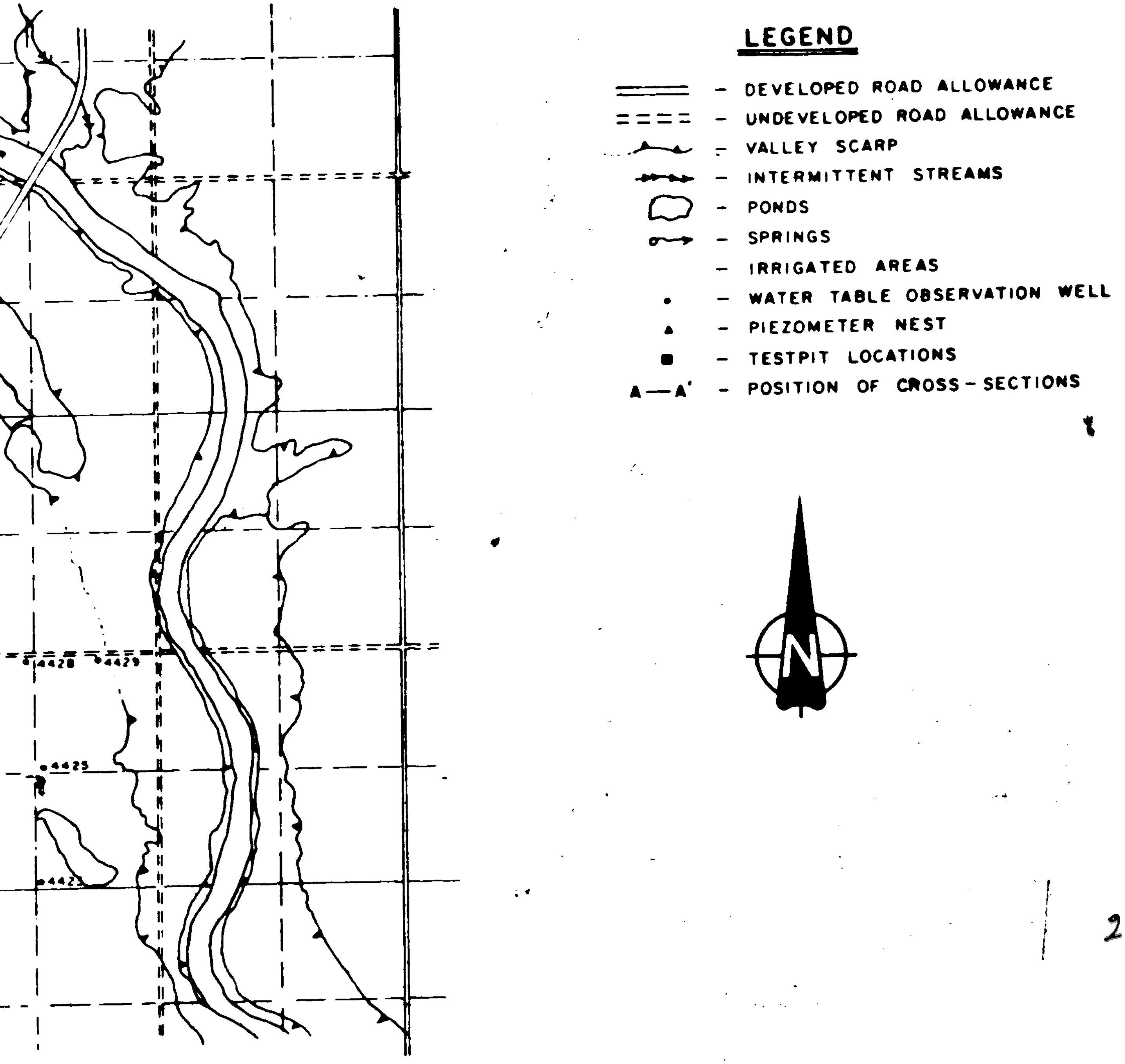

LEGEND

═══ - DEVELOPED ROAD ALLOWANCE
= = = = - UNDEVELOPED ROAD ALLOWANCE
▲ - VALLEY SCARP
➝ - INTERMITTENT STREAMS
◯ - PONDS
⟿ - SPRINGS
- IRRIGATED AREAS
● - WATER TABLE OBSERVATION WELL
▲ - PIEZOMETER NEST
■ - TESTPIT LOCATIONS
A—A' - POSITION OF CROSS-SECTIONS

2 of 2

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Alberta IRRIGATION
AGRICULTURE DIVISION

SUBMITTED
DATE
APPROVED
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TABER IRRIGATION RETURN FLOW STUDY

FIGURE 3 - SITE PLAN

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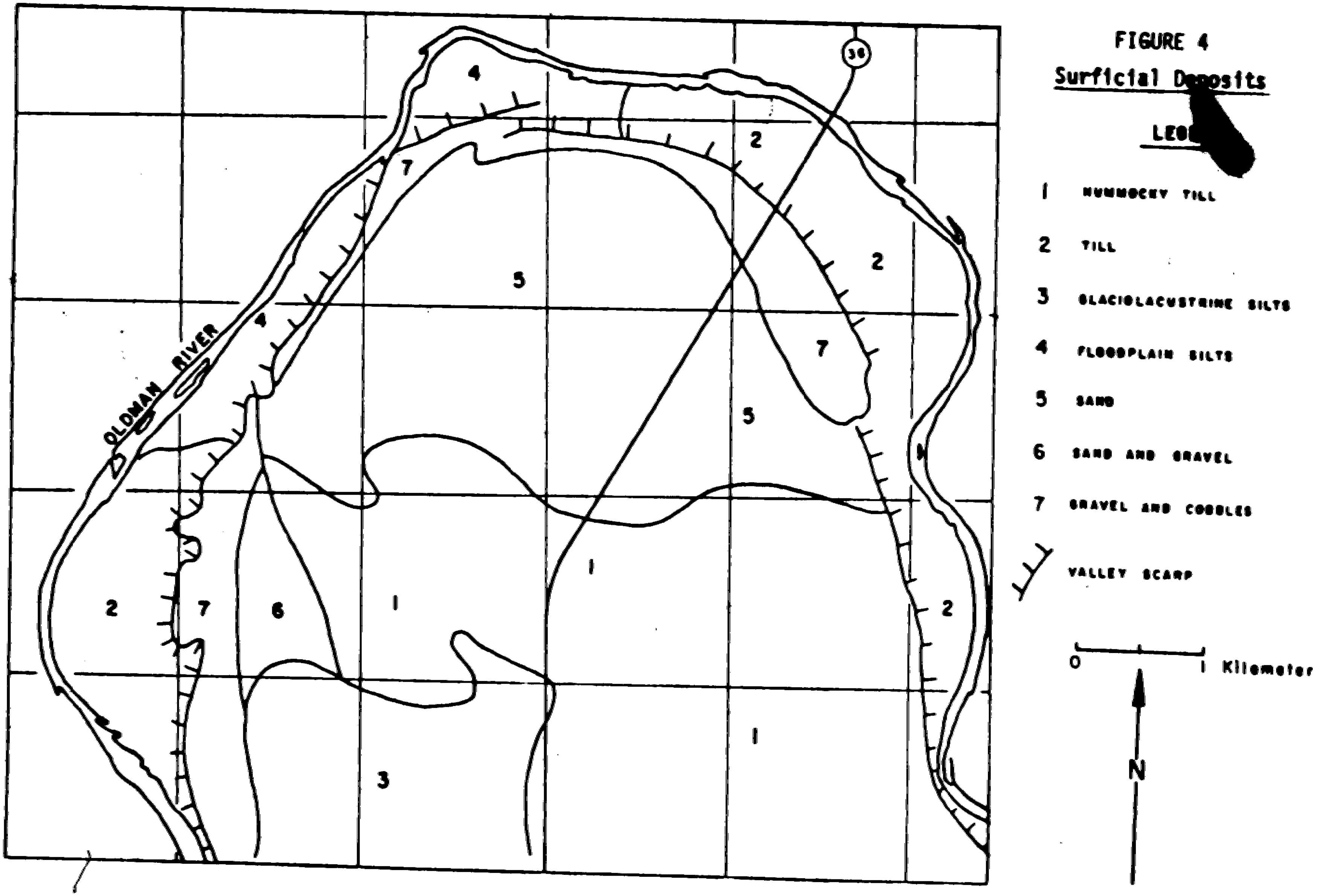
2. The unirrigated area which remains as natural grassland and is used for grazing beef cattle.

2.4 Overburden Deposits

The field area is underlain by glacially and fluvially derived deposits to a depth of 40-50m. Beneath these overburden deposits bedrock comprises the Upper Cretaceous Foremost Formation, a brackish water sequence of shales, lenticular sandstones and coal seams (Tokarsky,1974). The contact between bedrock and the overburden deposits is slightly undulating. Since this study is concerned with groundwater quality predominantly within the overburden deposits, the bedrock geology of the area will not be considered further. More detailed bedrock geology descriptions may be found in Nielsen (1971).

Extensive drilling by the Drainage Branch of Alberta Agriculture, Lethbridge, during the installation of piezometers and water table monitoring wells across the study site has provided a detailed geological picture of the overburden deposits. A plan view map of the overburden deposits and three generalized cross sections are shown in figures 4 to 7. The locations of these cross-sections are given in figure 3. Detailed borehole logs compiled during drilling of pilot holes in 1978, are presented in the Appendix. The overburden deposits beneath the study site were seen, during drilling in the summer of 1980, to be

FIGURE 4
Surficial Deposits



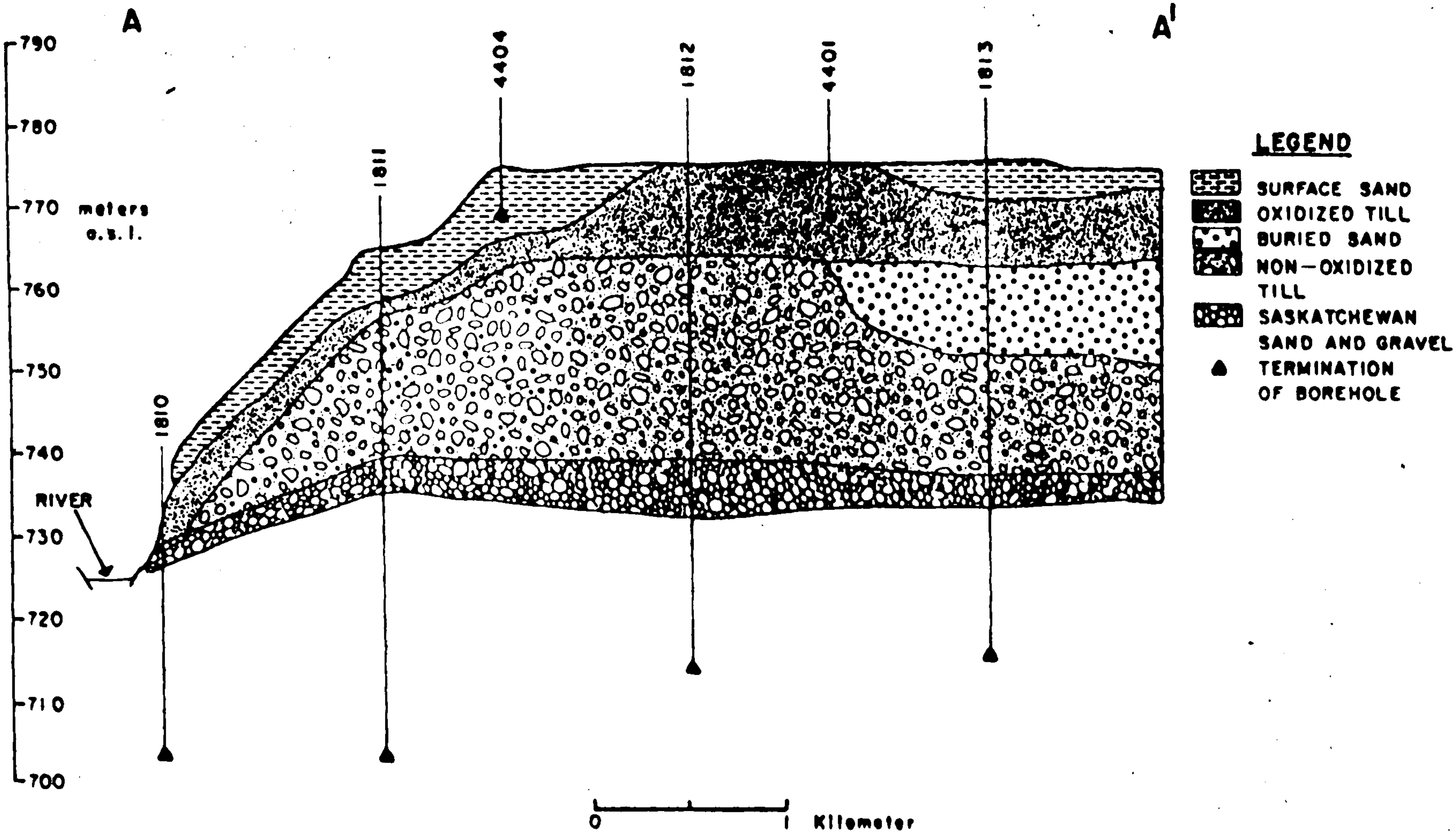


FIGURE 5: Overburden Geology Along Cross-Section A-A'

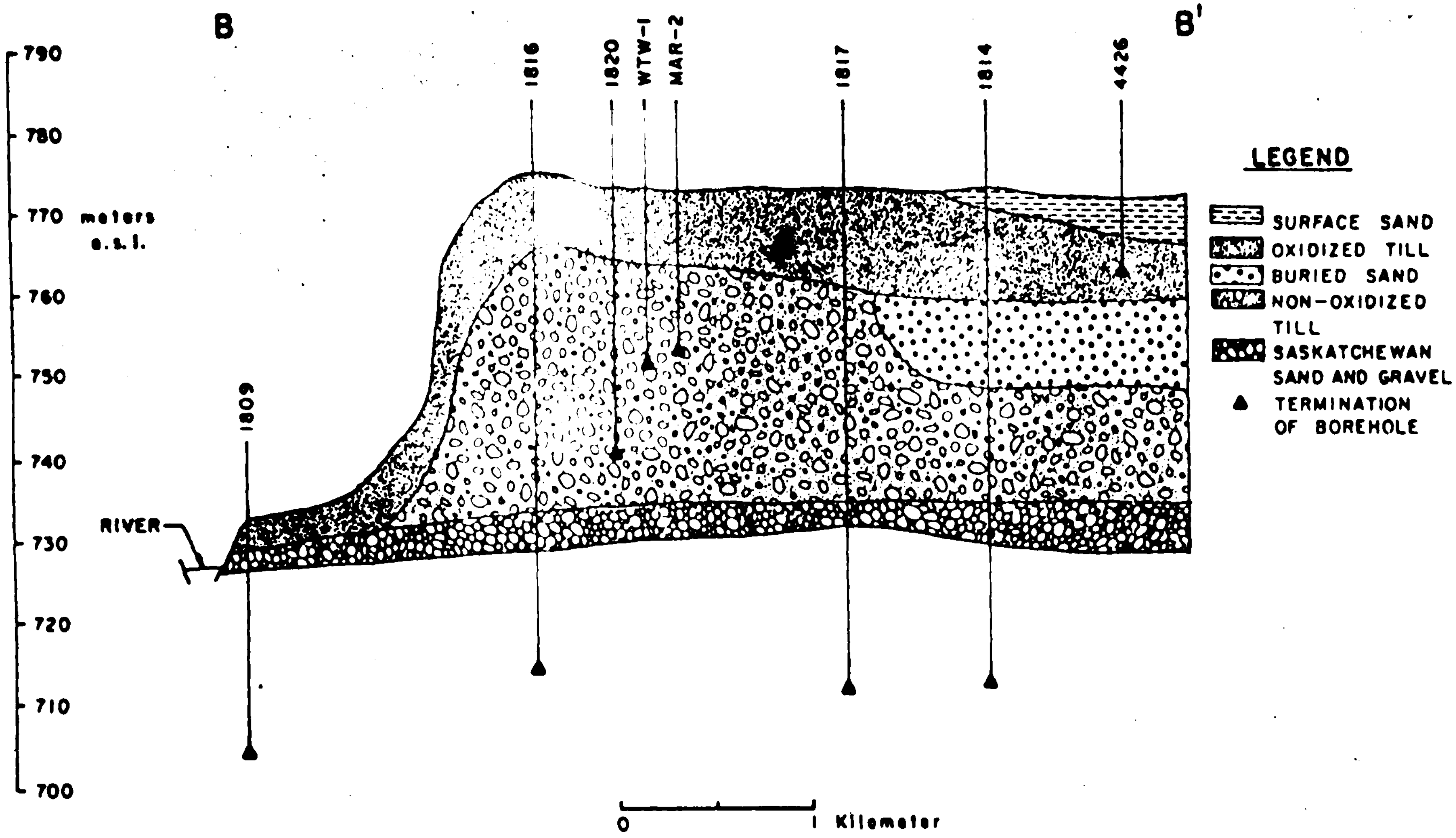


FIGURE 6: Overburden Geology Along Cross-Section B-B'

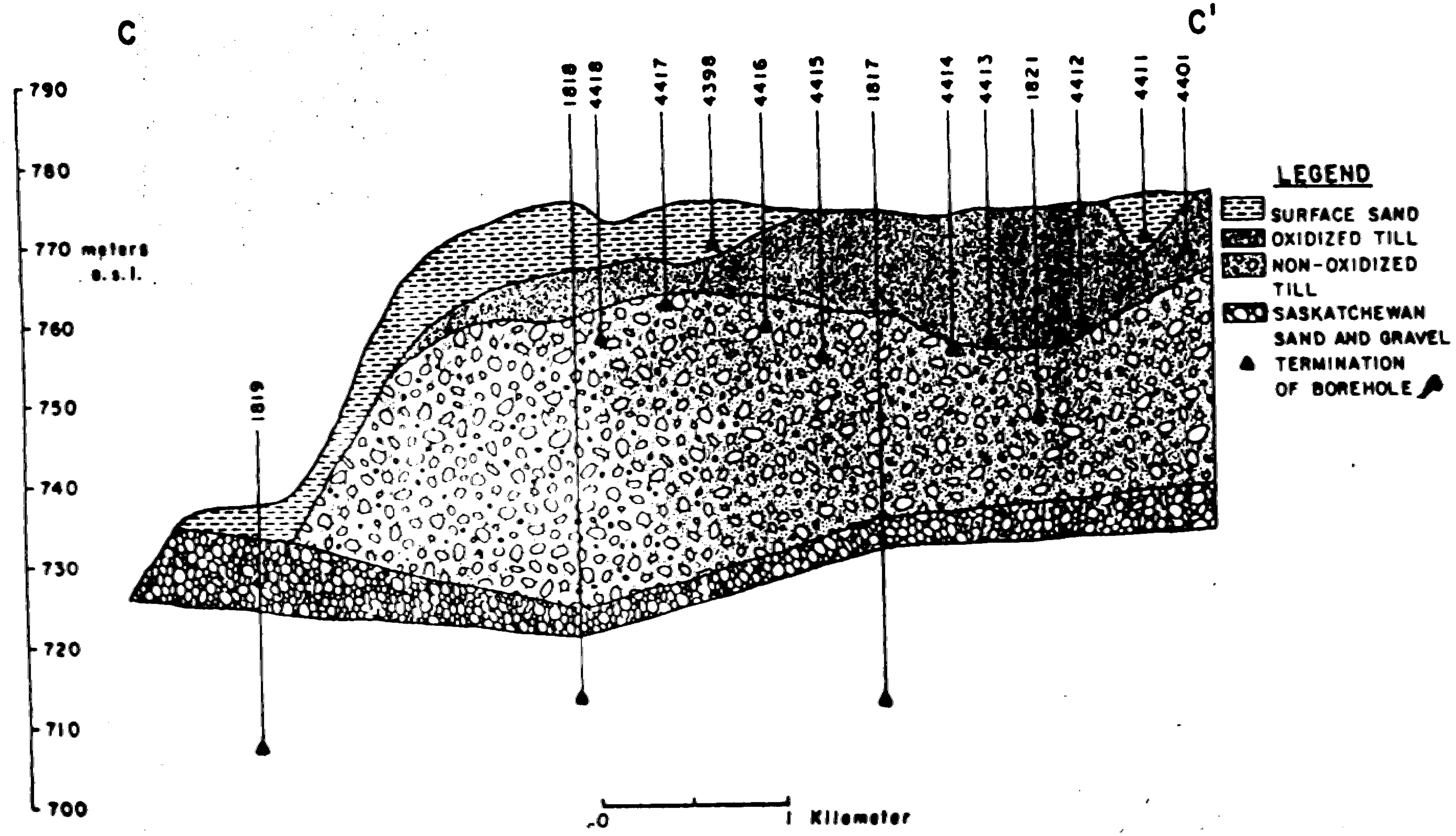


FIGURE 7: Overburden Geology Along Cross-Section C-C'

highly variable laterally as well as vertically. For example, at site 1813 one borehole was drilled which revealed 5 meters of surface sands however, another borehole 3 meters away on the surface revealed till at the surface. Such lateral variations in thickness of overburden geologic units was also found at depth within the deposits. Some inconsistency may therefore be found between generalized cross-sections and the detailed borehole logs presented in the Appendix. Where the observations made during the drill program of 1980 are inconsistent with the results recorded in 1978 (Appendix), the units and unit thicknesses observed by the writer during the 1980 program are used. This is especially evident in the discussion where generalized profiles beneath each piezometer nest are presented with the water quality results from each site. The generalized cross sections are, however, felt to adequately represent the variations in geologic unit for the purpose of this study. Much more intensive drilling would be necessary to define the variations in unit thickness occurring beneath the study site in greater detail.

2.4.1 Saskatchewan Sands and Gravels

At the base of the overburden deposits the preglacial Saskatchewan sands and gravels disconformably overlie bedrock. These sands and gravels were deposited by fluvial activity within the broad easterly trending preglacial Lethbridge valley (Geiger, 1965) which lies directly beneath the study site. The Saskatchewan sands and gravels comprise chiefly quartzites, hard sandstone and Crowsnest volcanics (Stalker 1962). Clasts are subangular to well-rounded although the majority are sub- to well-rounded. Clast size ranges from 3cm to 15cm and the matrix comprises medium to coarse sand. The ratio of sand to gravel varies through the deposit and was probably controlled by fluctuating fluvial regime at the time of deposition. Secondary carbonate cementation and iron-oxide staining are evident close to the upper and lower contacts of the deposit.

2.4.2 Tills

A number of Pleistocene Laurentide glacial advances from the northeast buried southern Alberta's preglacial topography beneath a complex sheet of glacial deposits including tills, proglacial lake sequences of both lacustrine and deltaic origin, and moraine complexes. The number of Laurentide advances into the area seems to be a point of controversy. Stalker (1962) positively identifies three tills but advocates as many as five advances into the area by distinguishing five possible tills at a section just west of the study area on the west bank of the Oldman river, (Tp.11, Rg.16, Sect.19, NW1/4). However, Horberg (1952), identifies only three tills at his "Driftwood Bend" section on the northern flank of the study area.

The generalized cross-sections for the field area (figures 5 to 7) illustrate that it was not possible to distinguish the tills encountered during drilling to the same degree as Stalker or Horberg. Although there are two tills shown in these cross-sections, the features by which they are differentiated may be the result of variable post-depositional alteration within a single till unit; alternatively, two or more till units may have been masked. The two tills shown in the cross-sections are considered here as separate units since their distinguishing characteristics impart different hydrogeological properties which are of interest to this study, however, it must be emphasised that these are not necessarily two separate tills according to other conventions for classification of glacial deposits.

The lower till identified here is referred to as the non-oxidized till. It is a blue grey highly compacted till with texture varying from sandy clay loam to silty loam. Clasts are predominantly sandstones, quartzites and Precambrian Shield material, they are angular to subrounded and range in size from 0.5cm to 5cm. Locally derived coal fragments are very common throughout the till as are intermittent sand pockets and sand lenses.

The upper or oxidized till shows the same textural and compositional characteristics as the lower non-oxidized till; however, it is distinguished by its colour and the occurrence of fractures. The oxidized till is brown to dark brown due to staining by oxides, especilly in the vicinity of the fractures. Colourless crystals, possibly of gypsum, are found both along fractures and within the till matrix, these are abundant in the oxidized till and also occur to a lesser extent in the matrix of the non-oxidized till.

Two scales of fracturing were identified during the TID Return Flow Study (Alberta Agriculture, 1981), and are referred to as large and small scale fractures. The small scale fractures have a spacing of approximately 1cm and comprise three fracture sets which result in a nearly cubic form. The large scale fracturing is vertical, less regular or distinctive and occurs intermittently to give the till a columnar appearance.

2.4.3 Sand Deposits

Sands and gravels are found within the tills, and often comprise minor discontinuous lenses ; however, a major buried sand lens is located beneath piezometer nests 1813 and 1814 (figures 5 and 6). This is thought to represent a buried channel deposit which was found by site investigation to trend north-south beneath the study area and intersect with an east-west trending buried channel about 500 metres south of the site.

In the north and west parts of the field area surface sands overlie the oxidized till. These sands were probably glacially derived from outwash but have subsequently been reworked by the wind, since they have highly uniform grain size and contain very little silt. Dune shaped features can be distingushed on air photographs of the areas covered by surface sands.

Surface sand pockets also occur at other localities across the field area and show abrupt contacts with the surrounding till. These sands appear to fill depressions in the hummocky topography at the till surface and may be the result of wind action on post-glacial outwash deposits.

A thin post-glacial loess layer covers much of site to a depth of 1 metre, however, in most cases where the land is worked for agriculture it has been incorporated into the soil by ploughing.

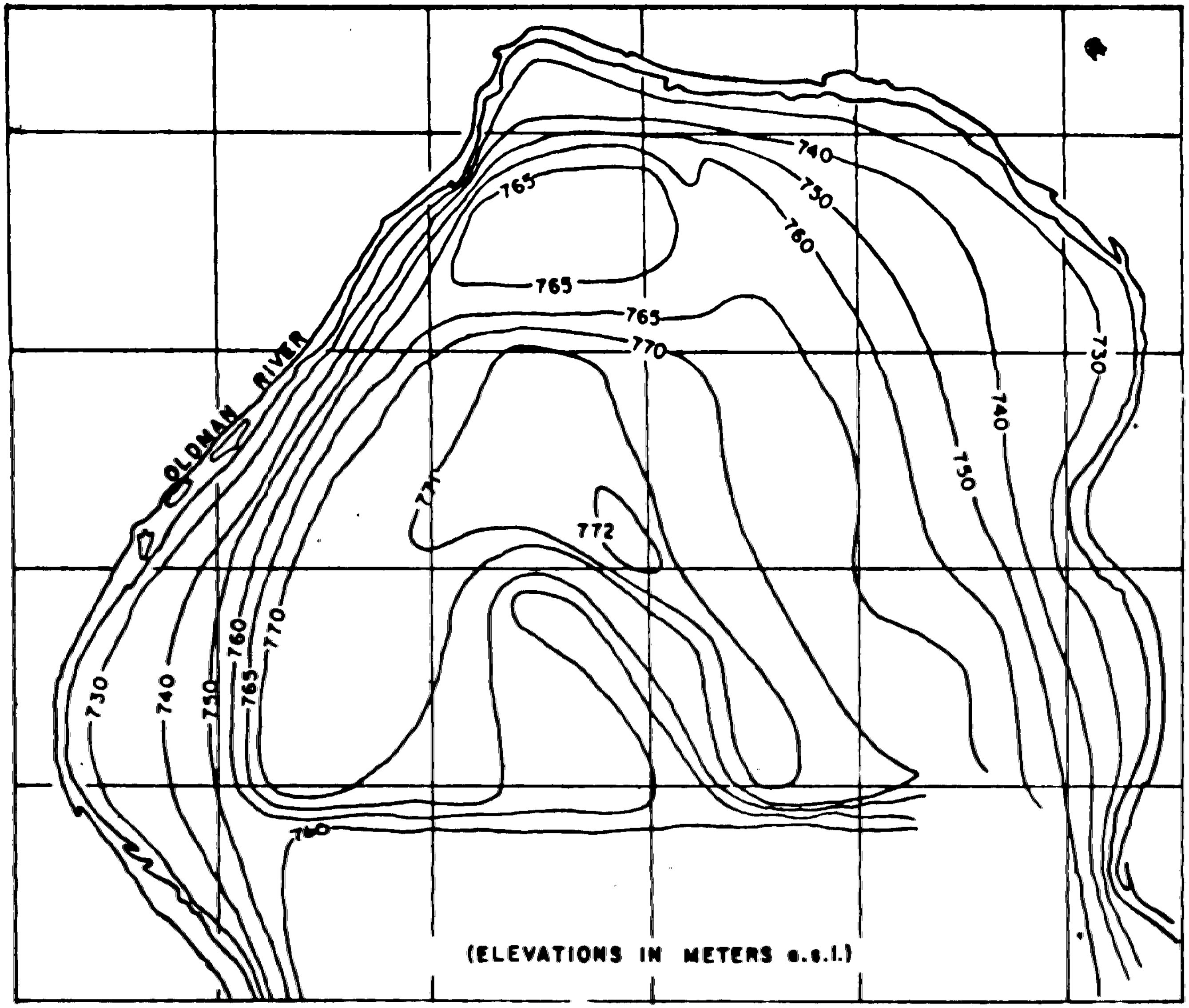
2.5 Hydrogeologic Properties of Overburden Deposits

Since this study is concerned with groundwater quality, it is important to establish the rates and directions in which water and solutes are transmitted through the overburden deposits. Modelling of the groundwater flow regime beneath the field area was included in the TID Return Flow Study (Alberta Agriculture, 1981). The following section is a summary of the results from the TID return flow study which are relevant to this study.

Observations and records of water table elevations and piezometric head readings across the site enabled construction of a water table map (figure 8), and cross-sections (figures 9 to 11), which show equipotential lines and generalized flow directions for the study site. Single piezometer water level response tests were also

FIGURE 8

Water Table Elevation



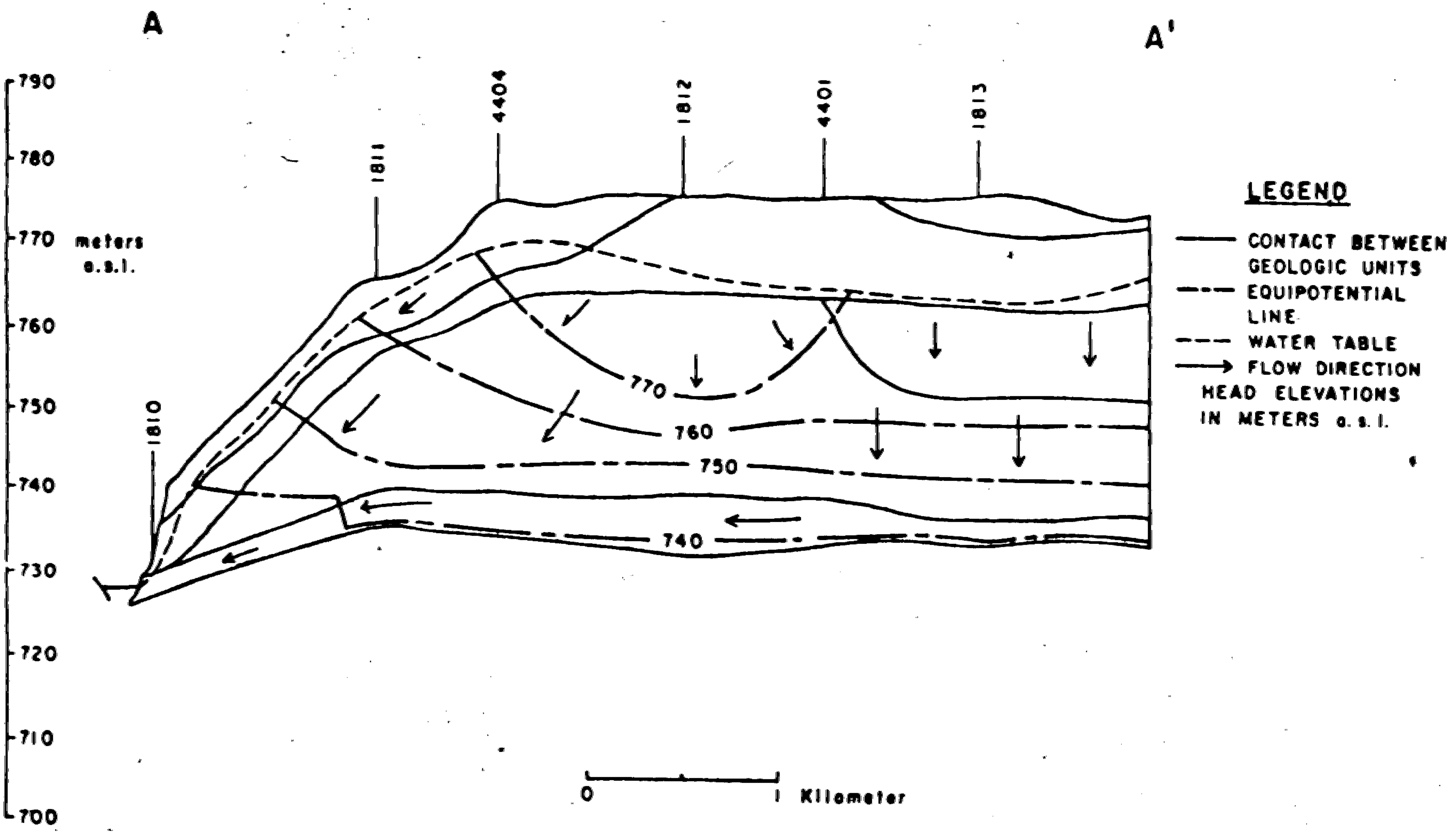


FIGURE 9: Equipotential Lines and Flow Directions Along Cross-Section A-A'

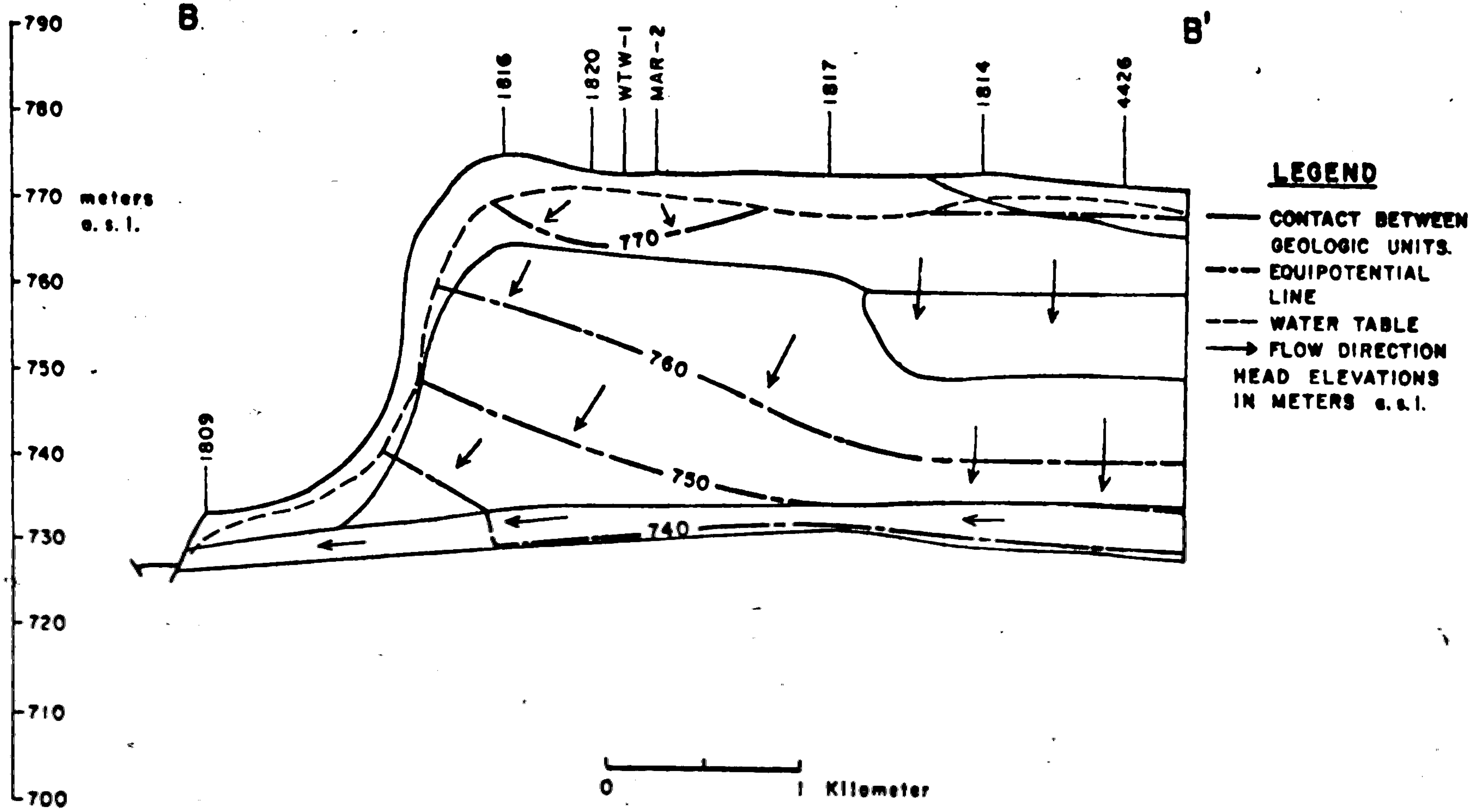


FIGURE 10: Equipotential Lines and Flow Directions Along Cross-Section B-B'

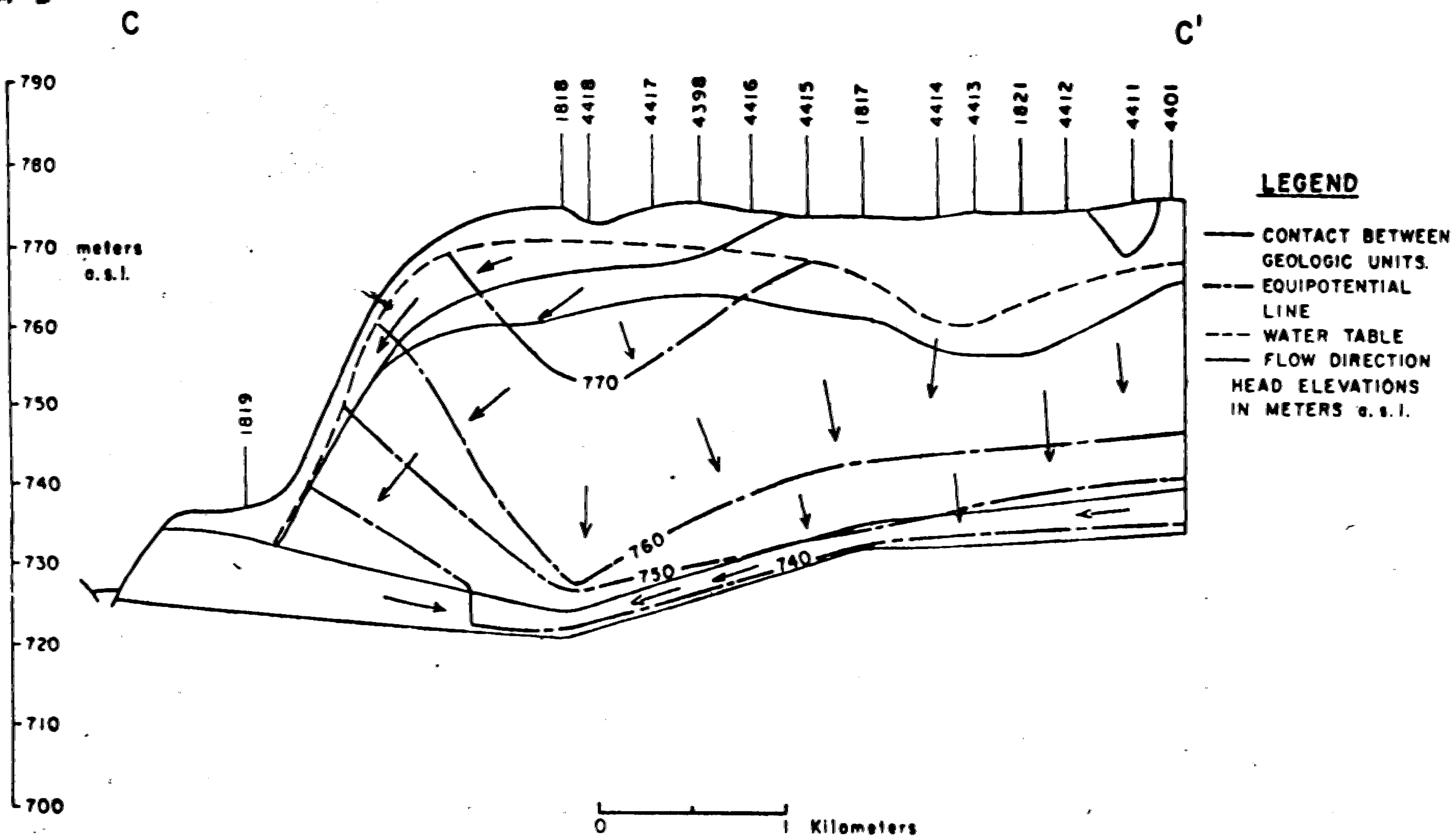


FIGURE 11: Equipotential Lines and Flow Directions Along Cross-Section C-C'

conducted to aid in determining hydraulic conductivities for the overburden deposits. Recharge to the water table was calculated as 10cm per year, which includes some irrigation water.

The hydraulic conductivities from the TID Return Flow Study are given in table 1. The surface sands have the greatest mean conductivities ($1.3\mathrm{x}10^{-3}$cm/sec), while the mean hydraulic conductivity of the buried sand is an order of magnitude lower at $1.1\mathrm{x}10^{-4}$cm/sec. The effect of the relatively high mean hydraulic conductivity of the buried and surface sands can be seen in figure 9. The buried sand lens beneath piezometer nest 1813 causes the flow lines to deflect toward it while the sands at the surface along the west side of the area drain water horizontally through them. Piezometer head measurements also indicate lateral flow from north to south within the buried sand lens beneath piezometer nests 1813 and 1814. Many of the sand lenses located during the drilling programme are laterally discontinuous, therefore, even though they are often saturated these sand lenses will have very little effect with respect to lateral movement of water or solutes within the till body.

The equipotential lines within the cross sections indicate predominantly vertical movement of water within the body of the tills, however rates of water movement vary. The upper non-oxidized till has a mean hydraulic conductivity of $7.9\mathrm{x}10^{-7}$cm/sec decreasing to $5.0\mathrm{x}10^{-7}$cm/sec with depth,

TABLE 1

Mean Hydraulic Conductivities from Single Well Response Tests

Unit	Range (cm sec^{-1})	Geometric Mean (cm sec^{-1})
Surface Sands	5.1×10^{-5} to 7.6×10^{-3}	1.3×10^{-3}
Buried Sands	6.3×10^{-6} to 1.1×10^{-3}	1.1×10^{-4}
Saskatchewan Sands and Gravels	6.5×10^{-3}	---
Oxidized Till	2.1×10^{-7} to 1.0×10^{-4}	7.1×10^{-6}
Non-oxidized Till	1.3×10^{-8} to 6.8×10^{-6}	5.9×10^{-7}
Upper	1.5×10^{-7} to 4.6×10^{-6}	7.9×10^{-7}
Lower	1.3×10^{-8} to 6.8×10^{-6}	5.0×10^{-7}

probably as a result of increased compaction. The mean hydraulic condutivity of the oxidized till is an order of magnitude greater than this ($7.1x10^{-4}$cm/sec). Since the matrix of the two till types is essentially the same with respect to grain size and texture, the greater bulk hydraulic conductivity of the oxidized till is probably the result of secondary permeability imparted by the fractures. This is corroborated by observations during drilling and sampling of overburden deposits, when films of water were seen along the fractures at points where the till matrix appeared only slightly moist. The hydraulic conductivities recorded here for the tills generally agree with those reported by Grisak et al. (1976) from a number of fractured till studies in the interior plains region of Canada and the United States.

The Saskatchewan sands and gravels have a greater hydraulic conductivity than the tills above due to their larger grain size and hence higher porosities ($6.5x10^{-3}$cm/sec), however, this only represents the result of one single well response test due to the difficulty in performing tests to the depths involved. Neilsen (1971) suggested that the hydraulic conductivity of the Saskatchewan sands and gravels may be highly variable since they are locally unsaturated. The hydraulic conductivity of the Saskatchewan sands and gravels is thought to be generally greater than that of the bedrock below, therefore the water movement within these deposits is shown to be lateral and toward bedrock depressions and the Oldman River. This may be important in the central part of the area (figure 10), since there appears to be hydraulic connection between the Saskatchewan sands and gravels and the Oldman river.

In summary the hydraulic conductivities of the overburden deposits beneath the field area vary through four orders of magnitude. Direction of water movement is predominantly vertical to subvertical, however some deviation from this pattern is observed within and close to the more permeable strata.

Rates of groundwater movement through the overburden deposits depend not only on the hydraulic conductivity of the transmitting medium but also on the porosity and the existing hydraulic gradient within the unit. Average flow velocities can be calculated using the Darcy equation which states that the average groundwater flow velocity (v) is the product of the hydraulic conductivity (K) and the hydraulic gradient (dh/dl) divided by the effective porosity of the transmitting medium (n). Hence, the average groundwater velocity increases with increasing hydraulic conductivity and gradient and decreases with increasing porosity.

Flow velocities calculated during the TID Return Flow Study (Alberta Agriculture, 1981) are presented in table 2. Porosities used in these calculations are taken from a similar study in the Bow River Irrigation District (BRID), (Hendry, 1981). The most rapid flow occurs within the

TABLE 2

Calculation of Average Flow Velocities from the Dupuit-Forchheimer Assumption

$$v = \frac{K \frac{dh}{dl}}{n}$$

i) Surface Sand

$K = 1.3 \times 10^{-3}$ cm sec^{-1}
$dh/dl = 0.08$
$n = 0.3$ to 0.4

$v = 2.2 \times 10^{-1}$ m day^{-1} to 2.9×10^{-1} m day^{-1}

ii) Oxidized Till

K fractures $= 7.1 \times 10^{-6}$ cm sec^{-1}
K matrix $= 5.9 \times 10^{-7}$ cm sec^{-1}
$dh/dl = 0.2$
n fractures $= 2 \times 10^{-3}$
n matrix $= 0.25$ to 0.4

v fractures $= 0.6$ m day^{-1}
v matrix $= 2.15 \times 10^{-4}$ m day^{-1} to 3.4×10^{-4} m day^{-1}

iii) Buried Sand

$K = 1.1 \times 10^{-4}$ cm sec^{-1}
$dh/dl = 0.02$ (vertical
0.004 (horizontal)

v (vertical) $= 4.75 \times 10^{-3}$ m day^{-1} to 6.3×10^{-3} m day^{-1}
v (horizontal) $= 9.5 \times 10^{-4}$ m day^{-1} to 1.26×10^{-3} m day^{-1}

iv) Non-oxidized Till

$K = 5.9 \times 10^{-7}$ cm sec^{-1}
$dh/dl = 1.4$
$n = 0.25$ to 0.4

$v = 1.5 \times 10^{-3}$ m day^{-1} to 2.4×10^{-3} m day^{-1}

fractures of the oxidized till because of the low porosity of the fractures. The surface sands have the second fastest flow velocities. The vertical flow through the buried sands is generally faster than the horizontal flow due to the difference in hydraulic gradient in the two directions. Flow is slightly faster through the matrix of the non-oxidized till than the oxidized till, due to the steeper hydraulic gradients in this part of the profile.

2.6 Hydrochemical Properties of Overburden Deposits

Since this study is concerned with the quality of the shallow groundwater, it is necessary to try to predict how the overburden deposits may change the chemical characteristics of the groundwater passing through them. The hydrochemical properties of the sands and gravels are quite different from those of the till, so they will be examined separately.

In a paper concerned with transport of radioactive contaminants through the groundwater system, Schwartz (1975) outlines two basic transport processes within groundwater: convection and dispersion. Convection is defined as chemical transportation as a dissolved phase by moving groundwater and it is assumed, at a first approximation, that the solutes travel in the same direction and at the same rate as the groundwater. The second physical transport process, dispersion, is known to accompany intergranular flow, hence

it is likely to occur in the sands and gravels beneath the field area. Dispersion is predominantly the result of mechanical mixing although some molecular diffusion may also take place, and this enables solutes to spread out, "beyond the region that they are expected to occupy according to the average groundwater velocity." (Grisak et al., 1976).
In this way dispersion is not just a physical transport process but also an attenuation mechanism. Dispersion occurs where the velocity of the contaminant front is greater or less than that of the convective front due to solute movement following longer or shorter flow paths (Schwartz, 1975).

The Saskatchewan sands and gravels and the outwash sands beneath the field area are expected to form the comparatively inert part of the groundwater flow system with respect to their effects on groundwater quality. Most of the clasts are composed of quartzite and quartzitic sandstones which are relatively inert under the conditions existing here.

The Saskatchewan sands and gravels and the outwash sands comprise clasts of fairly uniform grain size and contain very little clay size material, hence they are not expected to be the site of ion exchange reactions. Secondary cementation by calcium carbonate was observed within the Saskatchewan sands and gravels but it is uncertain whether this is the result of a contemporary process, or is relict from a former groundwater regime.

In contrast with the relatively inert nature of the sands and gravels, the tills present a much more complex environment with respect to their influence on chemical quality of the groundwater. Groundwater flow occurs predominantly along vertical fractures within the oxidized till which impart a bulk permeability that is an order of magnitude greater than the intergranular value. However, they also have a very low porosity so that flow through the fractures has potential to be extremely rapid (table 2). It is uncertain whether these fast flow velocities allow exchange of soluble species between the groundwater and the till matrix. If this does occur, it seems likely that any soluble material contained in the till matrix will initially be dissolved by the groundwater from the parts of the till closest to the fractures. This may set up an ionic diffusion flux down the concentration gradient from the interior of the till blocks toward the fractures so that progressively more and more soluble salts are removed from the till matrix by the groundwater (Grisak et al.,1976). However, diffusion in the opposite direction may also occur as documented by Day (1977) if concentrations of soluble species in the pore water are less than concentrations in the groundwater within the fracture network. Using a computer model, Day demonstrated that low ionic concentration gradients of less than 100mg/L/cm are capable of producing salt fluxes to and from the matrix. This diffusion of ions in and out of the matrix can alter the major ion content of water in the fractures whilst scarcely affecting the matrix water chemistry. Groundwater residence times in the oxidized till are short since flow in the fractures is rapid (table 2), therefore the degree to which the diffusion of salts in and out of the till matrix can occur beneath the study site, is uncertain.

The fractures are also the zones of concentration of oxidized minerals which, from the colour of the oxides, appear to be dominantly iron or magnesium. Hydrous oxides of iron and magnesium are known to have considerable adsorptive powers with respect to heavy metals (Gadde and Laitinen, 1974), and so these could act as retardants to the transfer of elements such as cobalt, nickel, copper and manganese through the groundwater system. Sulphate can also be attenuated in this way (Ensminger,1954;Harward and Reisenauer,1966)

A further property of the fractures is that they limit the degree of lateral dispersion of groundwater within the oxidized tills, however, longitudinal dispersion can occur along the fractures. Attenuation of solute concentration by dispersion is therefore expected to be minimal within this unit. Little else can be discussed concerning the effect of dispersion within the fractured tills since research into dispersion in fractured media is at present very limited.

Groundwater movement within the non-oxidized unfractured till is slower than in the oxidized till, allowing longer periods of contact between the groundwater and minerals within the till matrix. Water movement through the intergranular pore matrix also produces greater surface areas of contact between the till constituents and the water so that the chances of soluble minerals being dissolved are increased and dispersion can occur.

Both the oxidized and the non-oxidized tills contain high proportions of clays and clay-size material. During an investigation into south Saskatchewan till hydrochemistry, Rozkowski (1967) used X-ray diffraction techniques to establish that the clay minerals present in the tills were montmorillonite, illite and kaolinite. Montmorillonite comprised approximately 70% of the total clay minerals. Numerous other studies have also identified montmorillonite as the major clay within the tills of the Plains region. For example Hendry (1981) identified montmorillonite, (which is now known as smectite), as the major clay mineral, (50% of the total clay minerals present), in the tills of the Bow River Irrigation District (BRID) just north of the study site; Moran et al.(1978a) also identified smectite as the dominant clay mineral in tills from North Dakota.

Clay minerals are often the sites of cation exchange because of excess negative charges within their lattice structure. This process involves the exchange of cations held near the surface of an exchange mineral by a negative electrical charge, with dissolved cations in solution (Robinson, 1962). Since there are repeated exchanges between ions in solution and those held at the adsorption sites the adsorbed and soluble ions tend to exist in dynamic equilibrium. The rate at which the exchange occurs and the proportion of ionic species adsorbed is dependent upon a number of factors, including the concentration of the ion in the solution phase both singley and with respect to competing ions, the pH of the solution in which the ions are contained, and the cation exchange capacity of the adsorbing clay. The cation exchange capacity is expressed in terms of milliequivalents of exchangeable ions retained by 100g of the exchange medium. Hausenbuiller (1972) gives representative cation exchange capacities for common exchange minerals at pH7 as 2-16 meq/100g for kaolinite, 20-40 meq/100g for illite and 60-100 meq/100g for montmorillonite. Hence montmorillonite (smectite), which is probably the most abundant clay mineral in the tills beneath the study site has the greatest cation exchange capacity.

Not all cations are adsorbed by clays with the same force. Some cations are preferentially adsorbed over others because of difference in valence and radius of the ion. Since the force of attraction of a cation is not a fixed value, but varies according to the nature of the adsorbing material, the relative adsorption affinities of cations are often used. These are found to conform to the series;

$$H^{+}>Al^{3+}>Ca^{2+}=Mg^{2+}>K^{+}=NH_4^{+}>Na^{+}$$

A comprehensive and detailed discussion of cation exchange processes can be found in Robinson (1962). However, from the preceding brief discussion it appears that cation exchange reactions are likely to have a major impact on the chemical composition of groundwater within the tills beneath the study site. Preferential adsorption of divalent over monovalent cations may result in an apparently greater concentration of the monovalent ions within the groundwater and the attenuation of divalent species as the water passes through the system. The effects of cation exchange reactions on the water quality beneath the field area will therefore be considered during the interpretation of field data later in this thesis.

The final property of the tills which is likely to have an effect on the groundwater quality is the presence of soluble salts within the matrix. The origin of these salts is still in question, however the major extractable ions in the till are sodium and sulphate, (see later section on 5:1 extract analyses). Sodium sulphate deposits within the till are documented throughout Alberta, Saskatchewan and Manitoba as well as the northern United States, (Grossman 1968). Cherry (1972) also reports chloride deposits in the tills of southern Manitoba. Cherry hypothesizes that salt rich brines were squeezed into the tills when ice loads were transferred, in part, to the pore fluids in the Paleozoic formations beneath them. As these brines from below travelled upward they encountered lower temperatures and pressures and, as a result, sulphates and chlorides were precipitated in the pores of the Pleistocene deposits.

Grossman (1968) outlines a hypothesis for the squeezing of sodium sulphate rich waters into meltwater channels which were depositing stratified drift in preglacial bedrock valleys. These sodium sulphate deposits are thought to have been preserved beneath subsequent drift. These are two possible mechanisms by which the soluble salts may have been deposited within the tills. However, in the study area sulphates dominate over chlorides as the major extractable ion. The effects of these soluble salts on the hydrochemical evolution of the groundwater will be discussed in a later section.

3. Methods of Investigation

This chapter will describe the methods used to determine the variability in the chemical quality of water beneath the study area. The investigation procedure can be subdivided into four parts;

1. Sampling of piezometers and water table wells.
2. Field geochemical measurements.
3. Laboratory analysis of water samples at the Alberta Environment Water Quality Laboratory in Lethbridge.
4. Processing of laboratory results into a more manageable form by computer using the WATEQF programme (Truesdell and Jones, 1974). These points will form the subheadings within this chapter.

The network of piezometer nests and water table monitoring wells across the study site was established by Alberta Agriculture during the two year period before this study. Further drilling was undertaken during June 1980 in order to extend the groundwater instrumentation network, by the Drainage Branch of Alberta Agriculture in Lethbridge. Table 3 describes the installations at the piezometer nests across the study site. A piezometer nest comprises a number of piezometers completed at various depths and a water table monitoring well. The location of piezometer nests (1809-1821) and water table wells (4393 to 4430 and WTW-1) at the site are shown in figure 3. The original numbering system allocated during drilling will be retained throughout this thesis to avoid unnecessary confusion.

Both water table wells and piezometers are constucted of PVC pipe of either 38mm (1.5inch) or 51mm (2 inch) internal diameter. Water table wells range from 4.6 metres to 18.3 metres deep and have intake zones over most of their length, which usually terminate approximately 0.3 metres below ground surface. Water table wells were installed in drill holes and backfilled with drill cuttings which were tamped down. The region around the top of the pipe was sealed with bentonite to prevent surface water from percolating down the side of the pipe. Pipes were capped and slotted above ground level to maintain atmospheric pressure within them.

Piezometers were installed at predetermined depths and have intake zones of 45cm (18 inches) long. Silica sand was used to form a 1 metre thick sand pack around the intake zones. A bentonite seal at least 0.15metres thick was installed above the intake zone to prevent downward percolation of water. The piezometer hole was then backfilled with drill cuttings sealed at the surface, and slotted and capped in the same manner as the water table wells.

TABLE 2

Description of Piezometer Installations

Piezometer	Depth (m)	Drilled By	Date	Screen
1809 - 1	10.4	A.A.	April 1979	F
2	8.2	A.A.	April 1979	F
3	5.5	A.A.	April 1979	F
4	3.5	A.A.	April 1979	F
1810 - 1	24.2	Env.-A.A.	June 1978	F
2	21.7	Env.-A.A.	June 1978	F
3	17.8	Env.-A.A.	June 1978	F
1811 - 1	24.5	Camfield	March 1979	W
2	22.1	Camfield	March 1979	W
3	17.4	Camfield	March 1979	W
1812 - 1	45.5	Env.-A.A.	June 1978	F
2	32.5	Env.-A.A.	June 1978	F
3	22.9	Camfield	March 1979	W
4	15.6	Env.-A.A.	June 1978	F
5	9.4	A.A.	April 1979	F
1813 - 1	42.7	Env.-A.A.	June 1978	F
2	36.7	Env.-A.A.	June 1978	F
3	30.2	Camfield	March 1979	W
24.1	24.1	All-Kind	June 1980	W
4	19.5	Camfield	March 1979	W
5	15.4	Camfield	March 1979	W
6	9.6	Env.-A.A.	June 1978	F
1814 - 1	45.0	Env.-A.A.	June 1978	F
2	32.0	Camfield	March 1979	W
24.8	24.8	All-Kind	June 1980	W
3	20.3	Camfield	March 1979	W
4	14.8	Camfield	March 1979	W
5	9.2	Env.-A.A.	June 1978	F
6.9	6.9	Double D	June 1980	W
6	6.2	Env.-A.A.	June 1978	F
4.4	4.4	Double D	June 1980	W
2.8	2.8	Double D	June 1980	W
2.3	2.3	Double D	June 1980	W
1815 - 1	18.2	Camfield	March 1979	W
1816 - 1	41.6	Env.-A.A.	June 1978	F
31.9	31.9	All-Kind	June 1980	W
2	28.6	Env.-A.A.	June 1978	F
3	13.2	A.A.	May 1979	F
4	7.6	A.A.	April 1979	F
6.7	6.7	Double D	June 1980	W
4.5	4.5	Double D	June 1980	W
1817 - 2	41.8	Env.-A.A.	June 1978	F
3	36.8	Env.-A.A.	June 1978	F
4	32.5	Camfield	March 1979	W
5	26.7	Camfield	March 1979	W
21.8	21.8	All-Kind	June 1980	W
6	16.4	Camfield	March 1979	W
7.4	7.4	Double D	June 1980	W
7	7.0	A.A.	April 1979	F
5.7	5.7	Double D	June 1980	W
1818 - 1	52.9	Camfield	March 1979	W
42.6	42.6	All-Kind	June 1980	W
2	27.7	Camfield	March 1979	W
3	13.8	Camfield	March 1979	W
4	10.6	Env.-A.A.	June 1978	F
5	7.1	Env.-A.A.	June 1978	F
6	5.9	Env.-A.A.	June 1978	F
4.5	4.5	Double D	June 1980	W
3.5	3.5	Double D	June 1980	W
1819 - 1	25.2	Camfield	March 1979	W
1820 - 34.1	34.1	All-Kind	June 1980	W
27.1	27.1	All-Kind	June 1980	W
21.7	21.7	All-Kind	June 1980	W
5	7.5	Double D	June 1980	W
1821 - 27.0	27.0	All-Kind	June 1980	W
20.9	20.9	All-Kind	June 1980	W
15.0	15.0	All-Kind	June 1980	W

F = Fibreglass wrapped
W = Well screen

3.1 Sampling of Piezometers and Water Table Wells

Groundwater samples were collected from piezometers and water table wells on three occasions between May and August 1980, in order to identify chemical variability in the shallow groundwater. These water samples were subsequently analyzed for major chemical constituents at the Alberta Environment Water Quality Laboratory, Lethbridge.

In preparation for sampling, all standing water was removed from the piezometers and water table wells with a bailer so that fresh formation water could enter the pipes. Some installations responded too rapidly to allow removal of all standing water, and in these cases standing water was extracted from the piezometer pipe just prior to sampling.

Since bailing was such a long process, especially for the deep holes which contained large volumes of water, quicker methods of water removal were considered. Peristaltic pumps were not effective to the depths required for the study and introduction of compressed air to force out the standing water had previously been attempted by Hendry (1981). This was discontinued for fear of aerating the sand packs around the piezometer tips which would reduce permeability and alter redox potential. It was important to avoid disturbing the intake zones since the piezometers sampled during this study were also being used to measure hydraulic conductivities of transmitting media by single well response tests. Bailing was therefore found to be the only practical method for removing standing water.

A number of precautions were taken while using the bailer so as to minimise contamination of formation water in the installations both from the surface and between installations.

1. The bailer and rope were washed thoroughly with distilled water between sampling one installation and the next.

2. The bailer and rope were never placed directly on the ground surface.

3. The bailer was emptied away from the top of the installations.

After bailing the installations were left for a few days to allow them to recover and fill with fresh formation water. The installations which responded rapidly were sampled within a day of bailing.

Once the installations had responded and contained sufficient fresh formation water, samples were taken for laboratory analysis and field tests were performed. In order to obtain a laboratory sample, a sterilized 250mL polypropelene sample bottle was washed twice with distilled water and three times with formation water. The sample bottle was then filled with formation water from the bailer and capped tightly. The free space within the sample bottle was minimized in an attempt to prevent degassing and oxidation of the sample during transportation. The sample was labelled and immediately placed in a refrigerator. In some cases where samples were dirty, field filtering was attempted but this was too slow to be practical since much of the sediment in the water samples was very fine. The inclusion of sediment in the water samples may have induced slight changes in chemical quality of the water during transportation and storage. However, the sediment in the bottles was derived from the formations through which the groundwater had been flowing and so was not considered to cause any major changes in quality of the water samples.

Some investigators (e.g. Jackson, 1980), recommend acidification of samples with nitric acid to prevent precipitation of carbonates from the water sample during transportation and storage. This was attempted on a group of samples but no difference was found in analysis results between acidified and unacidified samples, so the practice was discontinued.

Within a day of sampling, samples were transported refrigerated to the Alberta Environment Water Quality Laboratory in Lethbridge where they were refrigerated until analysed. Analysis was usually performed within two days of sampling. Maximum storage time encountered between sampling and laboratory analysis was one week.

3.2 Field Geochemical Measurements

Since changes in water quality can occur during transportation and storage, a number of downhole water quality measurements were recorded simultaneously with laboratory sample collection. These included *in situ* measurement of temperature (degrees celsius), salinity (parts per thousand), conductivity (mhos), and dissolved oxygen (mg/L), as well as field determination of pH and redox potential,(Eh,mv.). *In situ* salinity, temperature and conductivity measurements were taken using a YSI Model 33 S-T-C-meter and YSI Model 331 probe with a 15m (50 ft) cable. Occasionally, water in the piezometers was too deep for the probe to reach so water was removed with the bailer and placed in a clean beaker in order that S-T-C measurements could be taken. The water sample was replenished periodically until readings stabilized. Consequently, the temperature readings for these samples may have been a degree or two higher than those taken *in situ*.

In situ measurement of dissolved oxygen were taken using a YSI 5700 dissolved oxygen meter and accompanying probe, (model YSI 57A). The dissolved oxygen meter was recalibrated two or three times each day since air temperature changes affect the probe sensitivity. A stable dissolved oxygen reading was usually obtained within 5 to 10 minutes. The accuracy of the dissolved oxygen probe and meter was 0.1mg/L.

Bailed samples were used for field measurement of pH and Eh since equipment was unavailible for taking measurements downhole. An Orion 407A/F Specific Ion Meter was used with accompanying relevant electrodes to measure both pH and Eh. For pH measurement a Fisher standard combination glass pH electrode was used. The meter and electrode were calibrated using pH7 and pH9 buffers.

Once the calibration procedure was complete, a bailed sample of fresh formation water was used to take a pH measurement. This was replenished every three minutes during pH measurement to prevent a significant temperature increase. During pH measurements, the glass electrode was immersed approximately 40mm into the sample which was stirred to eliminate any anomalous micro-environment around the electrode. However, stirring was not so fast as to allow streaming potentials to develop which could be detected by the electrode. Once the pH reading had stabilized, (between five and twenty minutes), it was recorded to the nearest 0.05 pH units.

Problems are often encountered during field pH measurements because of the fragility of the instruments being used, but it was hoped that these could be minimised during the study by following a rigid sampling procedure. Even so, some technical problems were encountered which prevented the measurement of field pH for the third group of samples.

Degassing of samples in contact with the atmosphere must also be taken into account when measuring field pH. This is increased by stirring the sample and may significantly alter its pH. Comparison of field and laboratory pH will be used later to show the effects of degassing. A second problem is encountered with dirty samples which may contain a high percentage of colloidal charged particles. These can set up electric potentials within the samples and give erroneous pH readings.

The Orion 407A Specific Ion Meter was also used to measure Eh. An Orion platinum redox electrode, (model 96-78), was used which has a reference electrode incorporated into its body. The elctrode/meter combination was checked for accuracy before and after each Eh measurement using a standard Zobell solution which has a calculable Eh at certain temperatures according to the equation:

$$Eh(Zobell)=0.185+0.00164(25-t) \qquad (1)$$

where t=temperature of the Zobell solution in degrees celsius, (Langmuir, 1971). The sample was replenished every five minutes with fresh formation water to prevent a significant temperature rise and to minimize oxidation. The Eh was recorded periodically until the readings stabilized (less than 5mv drift in 10 minutes). A stable Eh reading usually took between 40 minutes and one hour to obtain.

Some investigators including Hendry (1978,1981), have used flow cell techniques, for measuring Eh and pH in the field. Water is pumped from the piezometer, through an airtight chamber or 'flow cell' containing the sensing electrodes, and out again via an outlet pipe. This has the obvious advantages of minimising the contact of the formation water with the atmosphere and maintaining a flow of fresh formation water across the sensing electrodes.

Unfortunately, the flow cell technique could not be used in this study because the pump was unable to lift water from the depths encountered and many piezometers responded so slowly that they would soon be pumped dry.

A flow cell technique would have been a great advantage in measuring Eh since contact with atmospheric oxygen may cause oxidation of species which are present in their reduced state in the undisturbed formation water. Consequently Eh measurements recorded and presented in this study probably represent values of Eh greater than would be recorded if it were possible to measure Eh *in situ*. However, since a consistent measuring procedure was followed the results are assumed comparable and will be used to illustrate relative variations in Eh throughout the study area.

In order to assess the relative concentrations of salts within the deposits beneath the field area, bulk samples were taken from the sands and tills during drilling. Aqueous extracts from these samples were chemically analysed for major ions by the Alberta Department of Agriculture Soils Laboratory . The water extracts were obtained from air dried samples using the ratio of 1 part soil to 5 parts water by weight, following the procedure outlined by the U.S. Salinity Laboratory Staff (1954).

3.3 Laboratory Analysis of Water Samples

Samples taken from piezometers and water table wells were analysed at the Alberta Environment Water Quality Laboratory for all major anions and cations as well as pH and electrical conductivity (EC). A full analysis included total ionic concentration determinations for calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), boron (B), bicarbonate (HCO_3^{-}), chloride (Cl^{-}), sulphate (SO_4^{2-}), nitrate-nitrogen (NO_3-N), and ammonium (NH_4^{+}) ions. In some cases, where samples were not too dirty, manganese (Mn) concentration was also determined.

A number of methods were used to analyse the samples. Sodium and potassium concentrations were determined by a flame emission technique; calcium, magnesium, and manganese by atomic adsorption spectroscopy; and chloride and bicarbonate by titration. A Technicon AutoAnalyzer 11 was used to determine nitrate-nitrogen, sulphate, ammonium and boron concentrations using a colorimetric technique.

3.4 Computer Processing of Laboratory Analysis Results

Laboratory analyses gave values for total concentrations of anions and cations (mg/L) within each of the water samples, however it is often advantageous to know whether the water samples are saturated with respect to certain species, the molar ratios of pairs of species in the sample, the partial pressure of carbon dioxide with which the sample is in equilibrium as well as a number of other chemical variables. Determination of such variables would be very time consuming hence a computer program (WATEQF) was used to compute these variables and more.

The WATEQF programme was written by Truesdell and Jones and was described by them in a paper in 1974 where it was called WATEQ. However, the original PL1 version was modified with more recent thermodynamic data by Reardon (1977) who included a subroutine which considers isotopic forms of carbon. The programme has been renamed WATEQF to distinguish it from the original version.

WATEQF was chosen for use in this study since it was the most easily accessible computer programme of its type and is more than adequate for the scope of the study. With it the operator may calculate the equilibrium distribution of inorganic aqueous species in natural waters, using the results from the laboratory chemical analyses and *in situ* measurements of temperature, pH, and Eh (redox potential). The thermodynamic data used in the programme were compiled by Truesdell and Jones (1974) after careful consideration of availible experimental results. Useful output data include total concentrations of input species as both molalities and milligrams per litre (mg/L), an analytical and computed epm (equivalents per mille) balance to check the accuracy of the chemical analysis, distribution of species including single and paired ions, ratios between pairs of ions (e.g. Ca:Mg molar ratio), and level of saturation of the water sample with respect to a number of minerals.

The thermodynamic and chemical theory which forms the basis of WATEQF's calculations will not be documented here, however a detailed description of the equations and constants used in WATEQF can be found in Truesdell and Jones (1974). Wigley (1977) gives a good description of the thermodynamic background to a very similar British computer programme (WATSPEC) which can also be used to determine the equilibrium speciation of aqueous solutions. Texts by Garrels and Christ (1965) and Stumm and Morgan (1970) detail the theory of equlibria in aqueous solutions.

A detailed description of WATEQF and documentation for the use of the programme, as well as examples of formatted input data and the corresponding output, are availible from the Systems Analyst, Alberta Agriculture, Lethbridge.

4. Results

Methods of water sample collection and analysis have already been described. This section will present the results of field and laboratory water sample analyses as well as data derived from computer processing of laboratory analysis results. Detailed discussion of the results will be presented in a later section. Only brief comments on the results will be made here.

Tables 4 to 6 summarize the results of field testing and laboratory analyses of the water samples taken from piezometers and the water table wells at these test sites, where the intake zone covered only the upper geologic unit. Concentrations of major ionic species were converted to mg/L by WATEQF. Results in these tables are grouped according to geologic unit in which the piezometer or water table well was completed and hence the type of deposit from which the water sample was taken. Results of analyses of water samples taken from the water table wells where the intake zone covered more than one unit, or where just a single water table observation well was installed, are given in table 7. These water table well samples do not represent point source samples and are often from very shallow depths, coming from the region just beneath the water table.

Unfortunately, pH could not be measured in the field during collection of the third set of samples, because of technical problems. Eh was measured only once during the field season since the process was very time consuming. Difficulty was encountered in obtaining deep samples from bedrock and the Saskatchewan sands and gravels, and very few samples from these units were analysed. Therefore data on the chemical characterstics of the water samples are grouped according to the four major types of geologic unit from which the samples were extracted. The groupings in order of increasing depth are;

1. Surface sands
2. Oxidized till
3. Buried sands
4. Non-oxidized till

A number of general comments can be made about the results before considering them with respect to the four groupings above. Water samples from sites 1811 and 1818 have lower concentrations of many of the major ions than samples taken from similar deposits at the other piezometer nests across the field area. Both sites have sands or sands and gravels as surface deposits and both are located close to the margins of the major irrigated area. Although site 1818 has a pivot directly north of it, the summer of 1980 was its first year of operation. These appear to be the only similarities between sites 1811 and 1818 which may distinguish them from other piezometer nests across the field area. The depth of the water table is no different at these sites than at other sites, hence different recharge

TABLE 4

Chemical Analysis Results for First Set of Water Samples

HOLE NUMBER	DEPTH (m)	FIELD pH	LAB pH	FIELD EC (mmhos)	LAB EC (mmhos)	FIELD TEMP. °C	DISS. O_2 mg/L	FIELD Eh (mv)	TDS mg/L	Ca^{2+} mg/L	Mg^{2+} mg/L	Na^{+} mg/L	K^{+} mg/L	Cl^{-} mg/L	SO_4^{2-} mg/L	HCO_3^{-} mg/L	NO_3-N mg/L	NH_4-N ppm	Mn ppm	B mg/L
Surface Sand																				
1814-2.8	2.8	7.9	8.3	3.35	4.53	10.0	1.0	150	3827	138	190	747	7.4	39	2181	520	2.3	---	0.1	0.45
1814-4.4	4.4	7.5	8.2	8.75	11.9	10.0	0.4	155	11497	401	498	2460	23	255	7272	583	5.4	---	<0.1	0.2
1818-6	5.9	7.65	7.7	0.38	0.56	7.0	0.5	138	506	58	21	41	3.5	3.5	51	327	<0.1	0.6	0.3	0.2
1818-5	7.1	7.6	7.8	0.45	0.58	7.0	0.75	120	525	62	21	44	3.5	3.5	66	325	<0.1	0.3	0.2	0.2
Oxidized Till																				
1820-N(WTW)	6.0	7.4	7.7	9.0	12.8	9.0	2.1	175	12941	439	334	2858	32	241	7060	715	126	---	0.4	0.3
1816-5(WTW)	6.1	7.2	7.7	7.5	11.3	12.0	3.6	125	10351	429	321	2322	28	304	6436	630	0.7	---	0.6	0.3
1814-6.9	6.9	7.45	7.7	7.0	10.1	8.0	0.3	135	9476	433	460	1821	27	255	5788	689	4.2	---	0.1	0.3
1820-5	7.5	7.05	8.0	8	11.5	9.0	0.7	185	11510	453	272	2460	34	216	5970	804	130	---	0.1	0.3
1816-4	7.6	7.15	7.5	5.5	8.1	8.5	1.4	40	7553	543	203	1550	23	124	4332	772	<0.1	---	5.6	0.4
1814-5	9.2	6.95	7.3	6.0	9.3	5.0	---	145	8916	481	266	1871	26	328	4904	1013	<0.1	0.8	2.7	0.3
1812-5	9.4	6.7	7.0	5.75	7.7	10.5	2.0	180	3705	445	235	1405	25	78	3953	1149	<0.1	1.6	14.0	0.6
1818-4	10.6	7.7	7.8	0.47	0.7	8.0	0.7	135	622	40	9.7	113	4.7	3.5	87	362	<0.1	0.7	0.2	0.4
Buried Sand																				
1817-5.7	5.7	7.35	7.9	8.0	11.2	9.0	1.75	125	8692	551	282	2124	25	284	4448	356	576	---	0.1	0.2
1817-7	7.0	7.1	7.6	6.0	9.5	6.0	1.2	145	7220	517	261	1568	28	149	3929	447	321	---	---	0.1
1817-7.4	7.4	7.2	7.8	6.75	9.8	8.0	1.4	95	7749	629	328	1649	25	149	4097	382	490	---	0.6	0.2
1814-4	14.8	6.95	7.3	3.2	4.7	5.0	---	5	4133	287	107	788	14	44	2036	857	<0.1	1.9	1.7	0.5
1813-5	15.4	7.35	7.8	2.45	3.5	10.0	4.8	165	3124	471	156	232	11	25	1878	331	20	0.3	0.2	0.2
1813-4	19.5	7.2	7.7	3.5	5.0	9.0	4.7	165	4464	547	189	538	21	46	2656	417	48	0.6	0.6	0.2
1814-3	20.3	7.0	7.3	4.3	6.2	3.0	---	-65	5621	382	170	1069	16	151	2968	860	<0.1	2.3	2.2	0.45
Non-Oxidized Till																				
1816-3	13.5	6.8	7.3	4.9	7.2	8.0	0.4	-135	6636	449	158	1333	21	46	3799	828	0.1	---	1.3	0.7
1818-3	13.8	7.6	7.8	0.78	1.1	8.0	0.5	135	943	34	7.3	218	5.5	7.1	184	485	<0.1	0.9	0.2	0.7
1817-6	16.4	6.75	7.2	5.25	7.8	7.5	1.1	200	6947	475	286	1230	30	67	3655	1137	67	---	---	0.3
1811-3	17.4	7.45	7.8	0.65	0.93	12.0	0.5	90	740	66	23	96	8.6	3.5	143	398	<0.1	---	0.6	0.3
1811-2	22.1	7.8	8.0	0.7	0.93	11.0	0.15	60	748	50	17	122	14	3.5	136	404	<0.1	---	0.6	0.4
1812-3	22.9	7.1	7.4	3.4	4.9	10.0	1.7	-25	4407	297	112	834	15	49	1960	1136	<0.1	2.5	1.2	0.5
1811-1	24.5	8.05	8.2	0.65	1.1	12.0	0.1	40	933	20	6.1	230	7.8	7.1	145	517	<0.1	---	0.1	0.5
1817-5	26.7	6.65	7.0	4.5	6.4	10.0	0.2	-25	5809	445	202	977	24	35	3011	1114	<0.1	---	---	0.4
1818-2	27.7	7.55	7.6	1.1	1.6	8.0	0.7	-40	1398	38	9.7	340	6.6	7.1	276	719	<0.1	1.0	0.2	0.7
1816-2	28.6	---	7.5	4.0	5.4	13.0	0.1	-135	4712	337	101	936	16	53	2613	645	1.1	4.5	---	0.5
1813-3	30.2	7.5	7.8	3.4	4.9	10.0	4.4	80	4399	405	164	671	17	39	2502	585	11	2.5	0.9	0.4
1814-2	32.0	7.15	7.5	3.0	4.3	3.5	---	-85	4407	339	123	664	14	27	2070	768	<0.1	2.4	0.7	0.5
1817-4	32.5	6.85	6.9	3.2	5.4	9.0	0.2	-110	4926	461	182	701	19	21	2368	1165	0.8	---	---	0.6
Bedrock																				
1815-1	18.2	7.2	7.6	3.2	4.6	10.5	0.2	-95	4252	427	157	607	19	60	2387	598	<0.1	---	4.2	0.5
1812-1	45.5	---	7.7	1.7	2.3	12.0	2.7	20	1854	58	28	444	8.6	18	692	606	0.2	1.3	---	0.6
1818-1	52.9	8.1	8.4	1.25	1.9	11.0	2.0	160	1792	36	8.5	448	10	3.5	275	1009	<0.1	1.5	0.2	0.5

TABLE 5

Chemical Analysis Results for Second Set of Water Samples

HOLE NUMBER	DEPTH (m)	FIELD pH	LAB pH	FIELD EC (mmhos)	LAB EC (mmhos)	FIELD TEMP. °C	TDS mg/L	Ca^{2+} mg/L	Mg^{2+} mg/L	Na^{+} mg/L	K^{+} mg/L	Cl^{-} mg/L	SO_4^{2-} mg/L	HCO_3^{-} mg/L	NO_3-N mg/L	B mg/L
Surface Sand																
1014-2.0	2.0	7.8	8.0	5.1	6.5	10.0	5447	234	252	1067	9.4	57	3304	523	0.4	0.2
1014-4.4	4.4	7.5	7.9	8.5	12	9.0	11562	357	451	2597	23	190	7349	583	3.3	0.2
1010-6	5.9	7.5	8.2	0.48	0.6	9.0	545	62	21	41	3.5	<0.1	61	326	0.3	0.1
1010-7(WTW)	6.1	7.7	8.2	0.49	0.5	9.0	515	52	33	32	3.1	<0.1	56	339	<0.1	0.1
1010-5	7.1	7.5	7.9	0.48	0.6	9.0	517	62	19	44	3.5	3.5	61	323	<0.1	0.1
Oxidized Till																
1020-N(WTW)	6.0	7.2	7.6	9.2	12.9	10.0	11059	409	310	2667	33	282	6404	729	122	0.2
1016-5(WTW)	6.1	7.2	7.9	6.5	11.1	8.0	10396	393	280	2460	19	195	6308	661	0.4	0.3
1014-6.9	6.9	7.4	7.7	7.0	10.2	7.5	9435	395	372	1947	28	245	5620	692	2.2	0.5
1020-5	7.5	7.0	7.4	8.0	11.6	9.0	9998	443	270	2345	35	234	5718	817	130	0.3
1014-5	9.2	6.7	7.6	6.5	9.5	8.0	8647	415	245	1897	27	300	4750	1004	0.3	0.2
1012-6	9.4	6.4	7.0	5.5	7.8	1.0	6942	403	236	1330	24	64	3737	1140	<0.1	0.6
1010-4	10.6	7.6	8.2	0.55	0.7	9.0	615	42	11	108	4.7	<0.1	88	361	<0.1	0.3
Buried Sand																
1017-5.7	5.7	7.3	7.9	8.0	11.1	8.5	8834	401	253	2124	24	207	5043	381	260	0.1
1017-7	7.0	7.2	7.9	6.75	9.5	8.0	7930	405	286	1719	29	142	4664	447	187	0.1
1017-7.4	7.4	7.1	7.8	8.25	9.9	8.0	8006	575	263	1694	25	149	4693	232	232	0.05
1014-4	14.8	6.8	7.6	3.2	4.8	7.0	4401	263	163	809	14	32	2262	858	<0.1	0.4
1013-5	15.4	7.4	7.9	1.1	1.6	8.0	1267	182	60	92	6.6	3.8	600	318	3.7	0.1
1013-4	19.5	7.0	7.7	3.5	4.9	6.0	4336	503	216	499	20	35	2613	406	43	0.2
1014-3	20.3	6.9	7.6	4.3	6.4	3.0	5678	339	210	1074	16	131	3055	854	<0.1	0.4
Non-Oxidized Till																
1016-3	13.5	6.8	7.2	4.7	7.2	9.0	6369	423	134	1315	21	53	3583	839	<0.1	0.7
1010-3	13.8	7.6	8.2	0.85	1.2	9.0	954	36	8.5	221	5.9	3.5	188	487	3.3	0.5
1017-6	16.4	6.8	7.5	5.25	7.8	8.0	6637	445	259	1221	30	57	3468	1151	66	0.2
1011-3	17.4	7.2	7.4	0.65	0.9	10.5	657	56	17	92	7.8	7.1	82	395	<0.1	0.4
1011-2	22.1	7.4	7.9	0.7	1.0	11.5	744	54	18	115	14	7.1	125	411	0.1	0.4
1012-3	22.9	6.8	7.3	3.6	5.3	2.0	4619	240	176	860	16	60	2051	1215	<0.1	0.5
1011-1	24.5	7.9	8.1	0.7	1.2	11.0	949	14	4.9	244	7.4	3.5	151	523	1.0	0.5
1017-5	26.7	6.7	7.3	4.6	6.5	11.5	6062	437	230	1007	25	32	3179	1144	<0.1	0.5
1013-3	30.2	7.2	7.8	3.6	5.2	9.0	4544	389	209	683	18	32	2651	530	25	0.4
1014-2	32.0	7.1	7.7	3.1	4.6	5.5	3949	283	177	706	14	14	1926	829	<0.1	0.5
1017-4	32.5	6.7	7.2	3.8	5.6	11.0	5223	439	226	752	19	18	2570	1199	0.4	0.7

TABLE 6

Chemical Analysis Results for Third Set of Water Samples

HOLE NUMBER	DEPTH (m)	LAB pH	FIELD EC (mhos)	LAB EC (mhos)	FIELD TEMP. °C	TDS mg/L	Ca^{2+} mg/L	Mg^{2+} mg/L	Na^{+} mg/L	K^{+} mg/L	Cl^{-} mg/L	SO_4^{2-} mg/L	HCO_3^{-} mg/L	NO_3-N mg/L	NH_4-N ppm	Fe ppm	Mn ppm	B ppm
Surface Sand																		
1814-2.8	2.8	8.1	4.5	5.7	12.5	4997	232	214	904	9.4	57	2987	512	0.1	0.2	1.3	0.3	0.4
1814-4.4	4.4	7.9	9.1	12	11.75	11565	441	462	2483	25	248	7301	603	3.4	1.0	0.6	1.4	0.1
1818-6	5.9	7.8	0.4	0.6	10.5	512	60	21	41	2.7	7.1	48	332	0.1	<0.1	0.3	0.1	0.1
1818-7(WTW)	6.1	7.9	0.4	0.6	10.0	489	46	33	30	2.3	7.1	29	342	0.1	<0.1	0.8	0.1	0.1
1818-5	7.1	7.7	0.4	0.6	10.5	514	62	19	44	3.1	7.1	82	298	0.2	<0.1	0.4	0.2	0.1
Oxidized Till																		
1820-4(WTW)	6.0	7.5	9.6	12.7	9.5	11922	419	308	2942	33	241	7108	750	120	0.1	1.2	0.5	0.2
1816-5(WTW)	6.1	7.7	7.2	10.9	8.5	10075	397	267	2352	27	191	6148	692	0.4	0.2	0.2	1.0	0.2
1814-6.9	6.9	7.7	7.1	10.1	7.75	9873	489	388	1998	30	276	5956	735	1.3	1.0	0.4	0.4	0.1
1820-5	7.5	7.4	8.5	11.5	10.0	8284	441	267	2575	36	234	6100	821	130	0.5	0.3	0.3	0.1
1814-5	9.2	7.1	6.8	9.4	9.5	8727	463	261	1846	27	326	4784	1019	0.1	0.5	0.4	5.4	0.3
1812-5	9.4	7.1	5.8	7.7	9.5	7244	441	248	1370	27	82	3914	1162	<0.1	1.4	1.5	14.3	0.6
1818-4	10.6	7.9	0.6	0.7	10.0	609	40	11	108	4.7	7.1	72	366	0.2	0.5	0.4	0.1	0.4
Buried Sand																		
1817-5.7	5.7	7.7	5.2	10.8	9.0	8432	561	275	2074	25	291	4294	388	825	1.8	3.7	0.3	0.1
1817-7	7.0	7.7	6.8	9.4	11.5	7419	519	267	1669	30	152	3953	445	382	0.5	1.5	0.3	0.1
1817-7.4	7.4	7.6	7.0	9.7	8.5	7551	627	288	1644	26	163	3939	386	479	1.2	0.3	0.6	0.1
1814-4	14.8	7.2	3.2	4.8	8.5	4226	274	114	818	14	46	2084	874	0.1	1.8	0.6	2.8	0.4
1813-5	15.4	7.9	0.7	1.0	8.0	841	106	36	55	4.7	7.1	326	298	14	<0.1	2.3	0.1	0.1
1813-4	19.5	7.5	3.5	5.1	7.0	4474	507	201	568	20	50	2656	425	47	0.1	0.2	0.4	0.2
1814-3	20.3	7.7	4.5	6.4	9.5	5698	287	241	1087	16	145	3050	871	0.2	2.0	1.3	2.3	0.4
Non-Oxidized Till																		
1816-3	13.5	7.2	5.0	7.1	8.5	6387	409	134	1333	20	53	3591	844	<0.1	3.3	2.1	1.4	0.6
1818-3	13.8	7.9	0.8	1.1	9.0	905	26	8.5	214	5.5	7.1	149	496	0.05	0.6	0.2	0.2	0.6
1817-6	16.4	7.3	5.5	7.7	12.0	7089	485	302	1241	34	71	3732	1450	75	<0.1	0.7	1.5	0.2
1811-3	17.4	7.7	0.7	0.9	8.0	794	84	22	96	8.6	14	163	405	0.1	0.9	6.8	0.6	0.4
1811-2	22.1	7.7	0.7	0.9	8.5	812	74	22	118	13	14	163	415	0.1	0.7	0.2	0.7	0.5
1812-3	22.9	7.5	3.2	5.2	10.2	4762	343	134	857	16	53	2123	1238	0.05	2.5	0.3	1.1	0.5
1811-1	24.5	8.0	0.8	1.1	10	918	16	6.1	230	7.8	14	120	523	0.1	0.7	1.1	0.1	0.6
1813-3	30.2	7.7	3.5	5.2	9.0	4548	431	174	690	18	50	2570	591	26	2.4	0.3	1.3	0.4

TABLE 7a

Chemical Analysis Results for First Set of Water Table Well Samples

HOLE NUMBER	DEPTH (m)	LAB pH	FIELD EC (mhos)	LAB EC (mhos)	FIELD TEMP. °C	DISS. O_2 mg/L	FIELD Eh (mv)	TDS mg/L	Ca^{2+} mg/L	Mg^{2+} mg/L	Na^+ mg/L	K^+ mg/L	Cl^- mg/L	SO_4^{2-} mg/L	HCO_3^- mg/L	NO_3-N mg/L	NH_4-N ppm	B mg/L
1814-7	6.2	8.0	8.25	12	4.0	---	120	11763	365	474	2605	3.9	177	7382	755	0.1	0.2	0.3
1817-8	5.7	7.6	7.25	10.4	8.0	2.9	100	7973	531	261	1871	24	227	4308	358	393	---	0.05
WTW-1	6.2	7.6	5.1	7.25	10.0	1.4	180	6380	447	26	1182	23	42	3842	498	86	0.2	0.2
4393	6.2	7.9	8.0	11.1	9.0	1.9	135	10914	407	379	2437	31	188	6820	652	0.9	0.1	0.2
4394	6.3	7.9	0.6	0.8	9.5	2.3	55	706	60	23	97	4.3	11	121	390	0.2	0.2	0.2
4398	6.3	8.0	1.7	2.5	8.5	1.8	100	2247	208	126	267	11	28	1230	373	3.2	1.2	0.2
4399	6.2	7.8	0.6	0.9	8.5	1.8	110	773	72	40	83	3.8	7.1	221	350	0.1	0.1	0.2
4401	6.2	7.8	9.5	13.7	8.5	2.8	150	12088	383	322	3219	29	326	8117	491	1.1	0.3	0.2
4403	6.2	7.9	0.6	0.9	9.0	1.6	140	699	80	36	55	3.9	7.1	202	315	<0.1	0.6	0.1
4404	4.9	7.8	0.6	1.8	9.0	3.7	110	1546	154	96	147	5.5	18	725	399	0.8	0.8	0.2
4411	4.9	7.2	2.0	3.5	9.5	0.5	110	3485	601	140	207	9.4	46	1974	507	0.3	---	0.5
4412	15.4	7.4	8.5	10.7	11.0	0.6	70	11148	409	474	2322	30	47	6964	905	0.5	---	0.2
4414	18.2	7.2	7.25	9.2	11.5	1.7	165	7568	605	419	1416	42	64	3814	731	479	---	0.2
4415	17.9	7.7	5.0	7.4	9.5	1.1	165	6547	453	244	1276	26	74	3890	471	112	---	0.3
4416	14.5	7.9	1.8	2.7	10.5	1.4	170	2190	192	140	244	4.7	18	1119	467	5.5	---	0.6
4417	12.0	7.8	1.1	1.5	9.0	0.3	-50	1144	138	42	126	7.8	3.6	443	383	<0.1	---	0.6
4418	4.4	8.1	0.45	0.6	11.5	0.6	-205(?)	540	42	8.5	90	5.1	<0.1	56	339	<0.1	---	0.2
4426	4.3	8.0	1.4	2.0	11.0	2.3	145	1432	128	158	57	5.8	46	413	589	35.2	---	0.2
4427	8.0	7.4	12.0	16.5	9.0	1.8	150	18164	421	528	4483	36	184	11239	1273	<0.1	---	0.6
4428	8.1	6.9	3.6	5.4	10.0	1.0	155	5087	471	236	694	20	18	2968	682	0.2		0.4

TABLE 7b

Chemical Analysis Results for Second Set of Water Table Well Samples

HOLE NUMBER	DEPTH (m)	FIELD pH	LAB pH	FIELD EC (mhos)	LAB EC (mhos)	FIELD TEMP. °C	TDS mg/L	Ca^{2+} mg/L	Mg^{2+} mg/L	Na^+ mg/L	K^+ mg/L	Cl^- mg/L	SO_4^{2-} mg/L	HCO_3^- mg/L	NO_3-N mg/L	B mg/L
WTW-1	6.2	7.0	7.9	5.1	7.3	10.0	6196	435	241	1163	24	42	3708	496	87	0.2
4393	6.2	7.5	7.7	8.0	10.9	10.0	10339	373	351	2322	31	117	6484	660	0.6	0.24
4394	6.3	7.6	8.0	0.6	1.0	10.75	728	44	19	126	3.9	11	130	394	0.2	0.15
4398	6.3	7.8	8.1	1.8	2.5	9.0	1673	52	28	248	9.4	21	937	374	3.2	0.22
4399	6.2	7.4	8.0	0.62	0.96	9.5	664	52	28	87	3.5	7.1	139	347	<0.1	0.15
4401	6.2	7.6	8.0	10.5	14	12.0	12621	379	323	3196	31	358	7781	481	73	0.17
4403	6.2	7.7	7.8	0.62	0.90	10.0	627	64	27	64	4.3	3.5	129	334	0.5	0.12
4404	4.9	7.6	7.8	1.3	1.9	10.0	1426	116	81	175	5.5	21	624	401	1.0	0.27
4411	4.9	6.8	7.2	2.7	3.2	10.0	2914	465	141	172	8.6	32	1619	475	1.6	0.39
4412	15.4	7.25	7.7	8.8	12.6	10.0	12677	409	562	2667	32	50	7973	984	1.1	0.34
4413	16.2	6.6	7.3	6.2	9.0	9.0	7311	573	423	1315	31	106	3429	1004	428	0.32
4414	18.2	6.8	7.4	6.8	9.3	11.5	7076	543	374	1416	42	64	3391	741	505	0.29
4415	17.9	7.4	7.8	5.0	7.5	9.0	6379	433	214	1287	26	85	3751	478	105	0.29
4416	14.5	7.4	7.8	1.75	2.6	11.0	2090	176	151	218	4.7	18	1033	476	13.2	0.37
4417	12.0	7.3	7.9	1.2	1.6	5.0	1183	132	47	131	7.4	10.8	497	384	0.3	0.27
4418	4.4	7.5	7.9	0.55	0.7	12.5	558	36	9.7	97	5.1	7.1	25.4	378	0.2	0.27
4419	17.7	---	7.4	0.72	0.78	9.0	565	58	15	67	5.1	7.1	31	383	<0.1	0.24

TABLE 7c

Chemical Analysis Results for Third Set of Water Table Well Samples

HOLE NUMBER	DEPTH (m)	LAB pH	FIELD EC (mhos)	LAB EC (mhos)	FIELD TEMP. °C	TDS mg/L	Ca^{2+} mg/L	Mg^{2+} mg/L	Na^+ mg/L	K^+ mg/L	Cl^- mg/L	SO_4^{2-} mg/L	HCO_3^- mg/L	NO_3-N mg/L	NH_4-N ppm	Fe ppm	Mn ppm	B mg/L
VTW-1	6.2	7.5	5.5	7.1	12.0	6136	407	248	1163	24	46	3855	506	86	<0.1	0.1	<0.1	0.12
4393	6.2	8.0	8.0	11.1	10.0	10595	397	355	2377	120	30	6628	687	0.1	1.0	0.5	0.3	0.1
4394	6.3	7.9	0.57	0.78	9.75	664	52	21	94	3.5	7.1	91	394	0.5	<0.1	0.7	<0.1	0.19
4398	6.3	8.0	1.65	2.4	9.75	1856	132	114	244	9.8	25	941	387	3.1	<0.1	0.4	0.2	0.24
4399	6.2	7.9	0.62	0.88	11.0	692	58	32	85	3.5	7.1	158	348	<0.1	<0.1	0.3	0.1	0.19
4403	6.2	8.0	0.62	0.9	9.0	759	82	38	69	4.3	3.5	230	332	0.4	<0.1	0.3	0.1	0.10
4411	4.9	7.5	2.8	3.1	9.75	2787	429	141	172	7.8	28	1537	471	2.1	0.1	0.2	2.6	0.38
4412	15.4	7.7	9.1	12.0	11.5	12294	443	521	2579	32	57	7685	975	0.9	0.1	0.2	0.7	0.19
4413	16.2	7.4	6.2	8.9	9.0	7188	567	401	1303	32	11	3338	1017	423	6.9	0.4	1.7	0.26
4414	18.2	7.6	6.5	9.2	11.0	7298	589	388	1379	41	64	3617	760	460	1.2	0.3	2.3	0.24
4415	17.9	7.8	5.7	7:5	10.5	6516	451	221	1297	26	85	3857	477	102	0.4	0.9	0.4	0.1
4416	14.5	7.8	1.8	2.5	11.25	2072	198	134	214	5.1	21	99	49	15	0.3	0.6	0.1	0.29
4417	12.0	7.8	1.3	1.6	8.5	1294	160	55	124	7.0	7.1	557	383	0.2	0.5	1.0	0.9	0.19
4418	4.4	7.9	0.52	0.65	11.5	564	38	11	94	4.7	<3.5	48	368	0.1	1.1	1.0	0.2	0.24
4426	4.3	8.0	1.6	2.1	11.0	1598	154	174	64	5.5	53	485	617	44	<0.1	0.2	<0.1	0.21
4427	8.0	7.6	12.5	16.6	10.5	17117	437	495	4198	36	191	10470	1289	0.7	2.3	0.5	18.7	0.5
4428	8.1	7.6	7.3	5.4	10.0	4966	441	221	713	3.9	21	2858	708	0.1	1.0	0.1	9.0	0.38
4430	6.1	7.7	---	2.0	---	1713	124	58	244	146	7.1	677	456	0.1	1.8	0.5	0.5	0.14

rates at the centre and the periphery of the irrigated area cannot be invoked. Leaky piezometer tips at either site are also unlikely to be a cause of the low observed ionic concentrations since hydraulic conductivities obtained by falling head tests on these piezometers (Alberta Agriculture, 1981) were generally comparible with those obtained from the other test sites. Only the 13.8m piezometer at site 1818 could be invoked as being leaky from the results of these tests.

When hydraulic gradients through the surficial deposits at site 1818 are computed from the head elevations presented in the TID Return Flow Study (Alberta Agriculture, 1981), net gradient is found to be downward. However, gradients at site 1818 are very small, being almost imperceptible at the scale of measurement. This indicates that the groundwater at site 1818 is moving only very slowly downward since flow through times are directly related to hydraulic gradient. Such long flow through times compared with at other sites across the study site may be responsible for the low ionic concentrations in the water samples taken at site 1818, although the mechanism by which this occurs remains unclear.

Hydraulic gradients at site 1811 are not nearly as low as at site 1818, but indicate net downward movement of water through the non-oxidized till to the underlying Saskatchewan sands and gravels. Hence, slow flow through times cannot be invoked to account for the low concentrations of ionic species in the groundwater from the non-oxidized till at site 1811.

Unfortunately the differences between groundwater samples from piezometer nests 1811 and 1818 and the rest of the field area were not realized until after bulk samples of the overburden deposits were collected for the 5:1 aqueous extract analyses, so that the chemistry of the deposits could not be tested as a possible cause of the observed differences. However, from observation of the deposits beneath site 1818 during the 1980 drill program, no significant difference was observed between the deposits here and at other piezometer nests across the field area. Two samples of the surface sands were taken at site 1818 but with the exception of low sodium concentration in their 5:1 extracts, which resulted in slightly lower ECs for the extracts from these samples, they were not significantly different from surface sand samples collected elsewhere in the field area. The results of the 5:1 analyses will be presented in detail later in this chapter. The reasons for the anomalously low concentrations of major ion species at sites 1811 and 1818 remain unresolved.

4.1 Electrical Conductivity of the Water Samples

The electrical conductivity (EC) of the water samples was measured both in the field and in the laboratory. Laboratory EC was found to be consistently greater than field EC. Figure 12 is a plot of the two types of EC

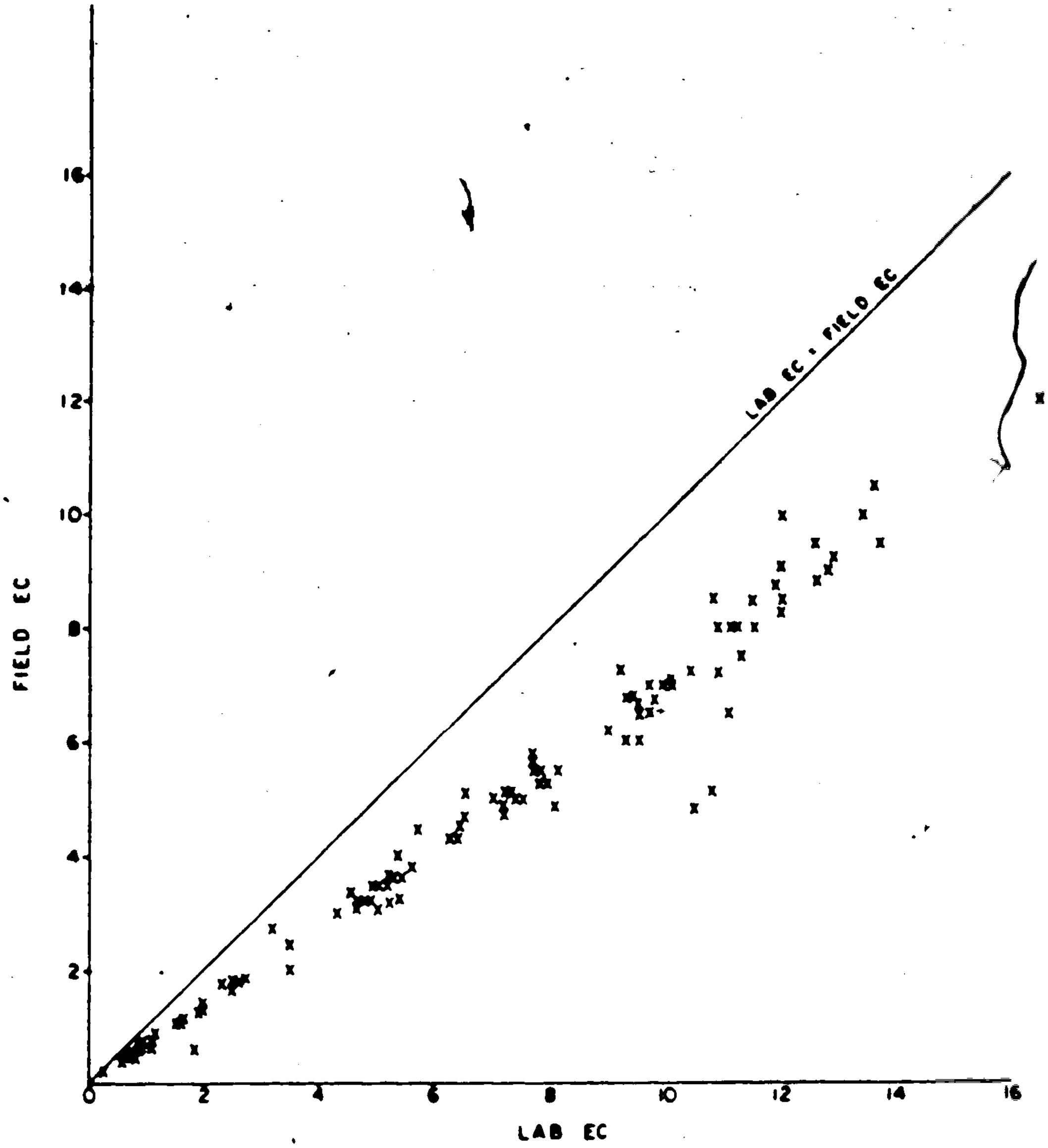


FIGURE 12: Field Electrical Conductivity of Water Samples versus Laboratory Electrical Conductivity.

measurement, which show a linear correlation. The equation describing the relationship is:

$$\text{predicted field EC}=0.71(\text{lab EC})+0.026 \tag{2}$$

with a correlation coefficient of 0.95 at a 95% confidence limit. Differences in the calibration of the two conductivity meters may be responsible for the observed difference between field and laboratory EC. Changes in the samples after collection seem unlikely to produce such a marked linear relationship between the field and laboratory measurements. Degassing of carbon dioxide should cause the laboratory EC to be less than the field EC however the reverse case was observed so that degassing effects after sample collection could not be invoked. Further investigation into the calibration of the two conductivity meters would be necessary in order to establish the reasons for these differences.

As expected, laboratory EC is positively correlated with the concentration of total dissolved solids (figure 13). The equation describing this relationship is ;

$$\text{predicted lab EC}=0.001(\text{TDS})+0.29 \tag{3}$$

with a correlation coefficient of 0.98 at a 95% confidence limit.

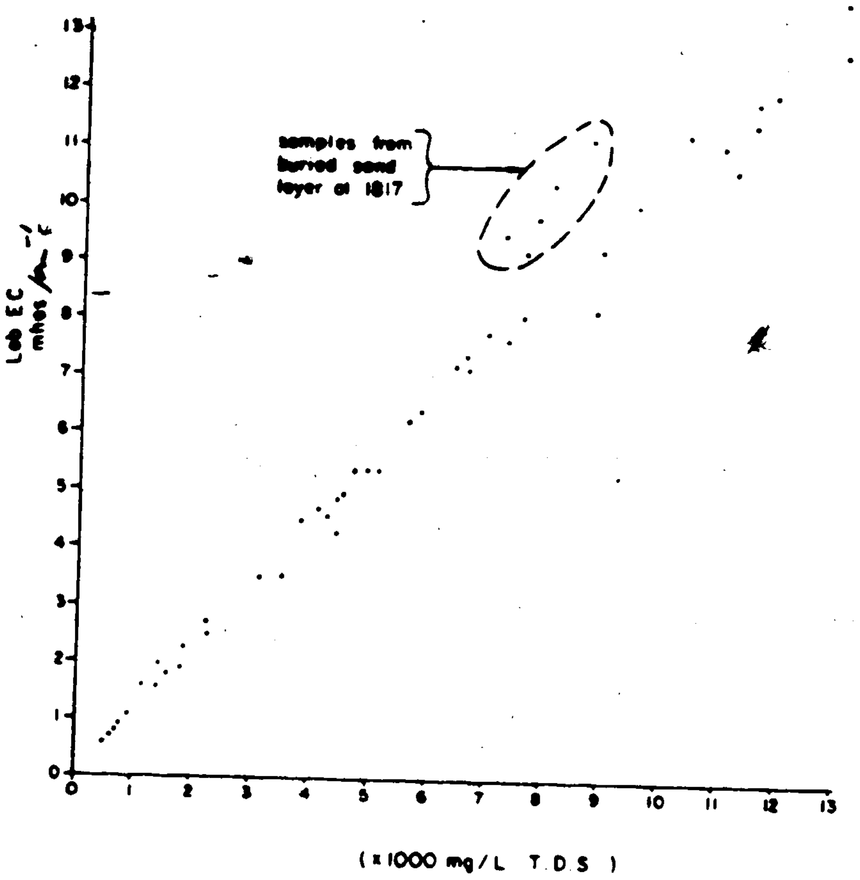


FIGURE 13: Laboratory Electrical Conductivity of Water Samples versus Total Dissolved Solids Concentration (1st Sample Set).

4.2 pH of the Water Samples

Field pH was measured during collection of the first two sample sets whereas laboratory pH was measured for all samples. Field pH ranged from 6.4 to 8.2 while laboratory pH ranged from 6.9 to 8.2. Laboratory pH is usually higher than field pH, although no correlation appears to exist between the two variables. Instead, the relationship between field and laboratory pH seems random (figure 14). The higher readings of pH in the laboratory than in the field probably result from the degassing of dissolved carbon dioxide between the times of sampling and laboratory analysis. The consequences of the degassing and its effects on the water chemistry will be discussed later.

4.3 Dissolved Oxygen in the Water Samples

Dissolved oxygen content of the groundwater samples was measured in the field during collection of the first set of samples, and concentrations were found to range from 0.1mg/L to 4.8 mg/L. Dissolved oxygen concentration was expected to decrease with depth since oxygen present in the shallow water is used by bacteria for respiration and in oxidation reactions as water percolates down through the profile. However, as figure 15 illustrates, this was not the case. Dissolved oxygen concentration is higher in two of the water samples taken from bedrock (2.7mg/L and 2.0mg/L), than in many of the samples taken from much shallower depths and is

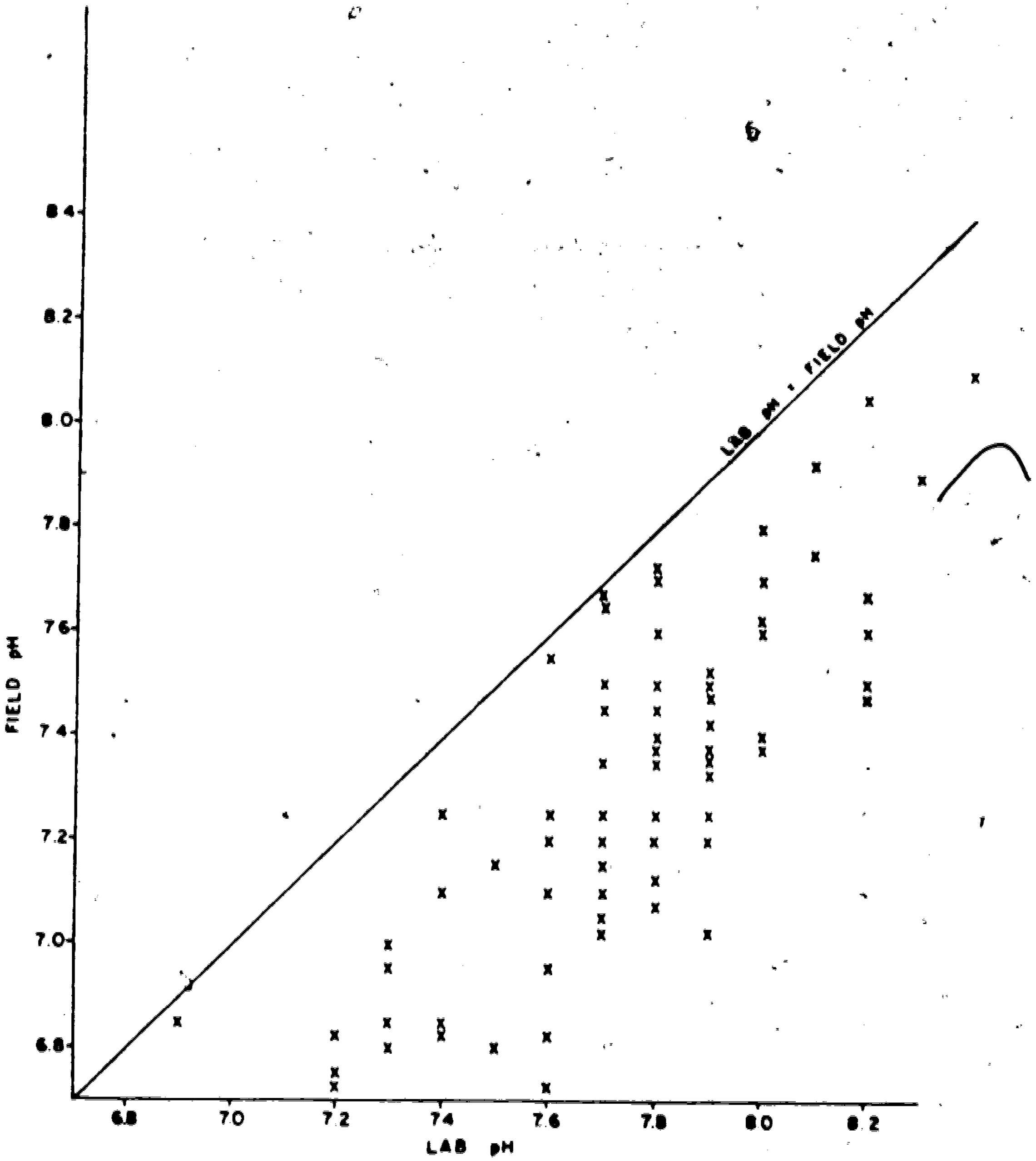


FIGURE 14: Field pH of Water Samples versus Laboratory pH (1st and 2nd Sample Sets).

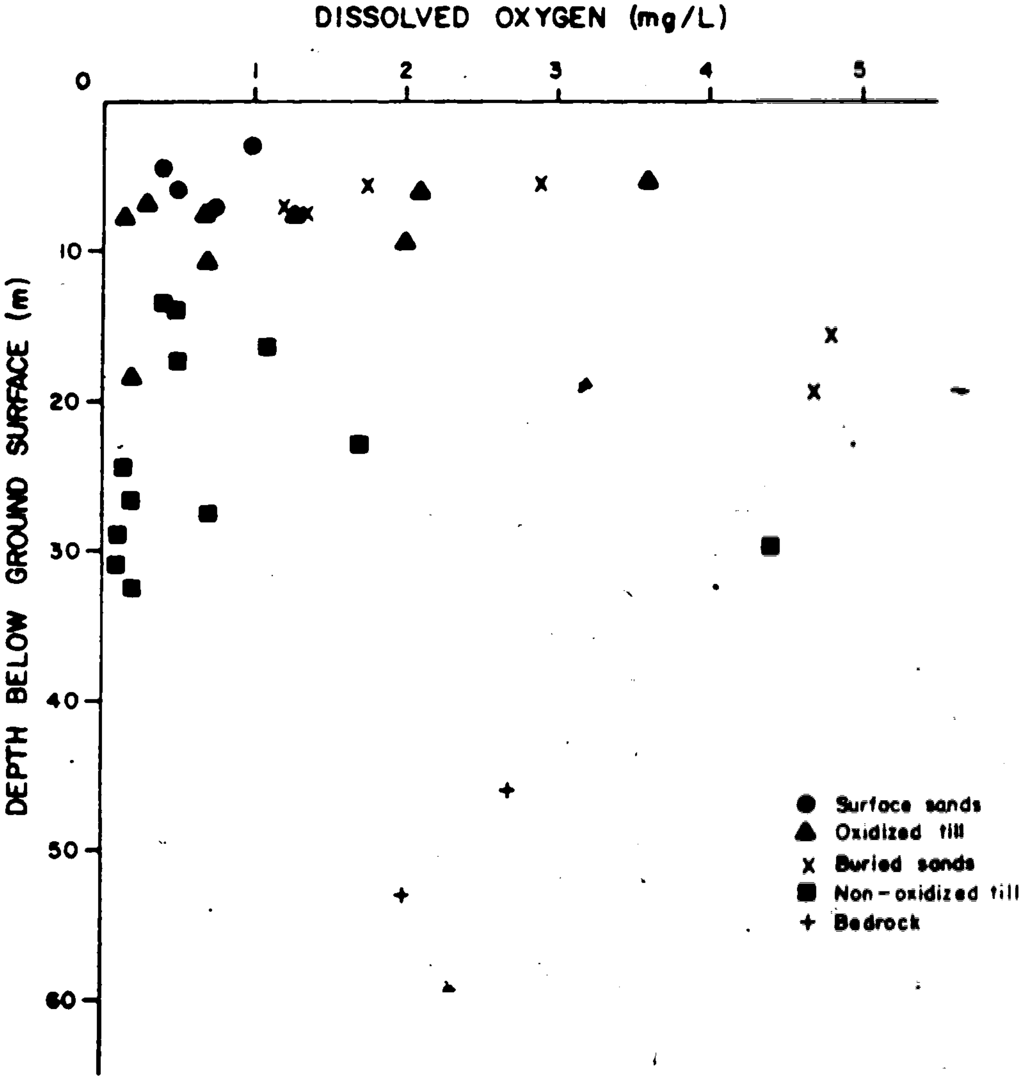


FIGURE 15: Dissolved Oxygen Concentration versus Depth from which Water Sample was taken (1st Sample Set).

also greater than all four samples taken from the surface sands. The bedrock at the depths where these two samples were taken was logged at the time of piezometer installation (1978) as brown siltstone and sandstone indicating that oxidizing conditions may exist at this depth since the brown colour is imparted by minerals in their oxidized state. The highest dissolved oxygen concentrations are present in the samples from the two piezometers in the buried sands beneath testsite 1813 (4.7mg/L and 4.8mg/L). The effects of dissolved oxygen concentration on the processes occuring in the groundwater flow system will be examined in detail during the discussion.

4.4 Redox Potential of the Water Samples

Redox potential Eh(mv), was also measured in the field during collection of the first set of samples. The results plotted against depth from which the sample was taken are presented in figure 16. In a similar way to dissolved oxygen concentration, Eh of the water samples shows surprisingly little correlation with depth. A plot of Eh versus dissolved oxygen for the first sample set is illustrated in figure 17. The expected linear trend between the two variables was not observed. This lack of linear correlation is probably the result of the inherent problems in measuring Eh in direct contact with the atmosphere and further illustrates the necessity of using the flow cell technique wherever

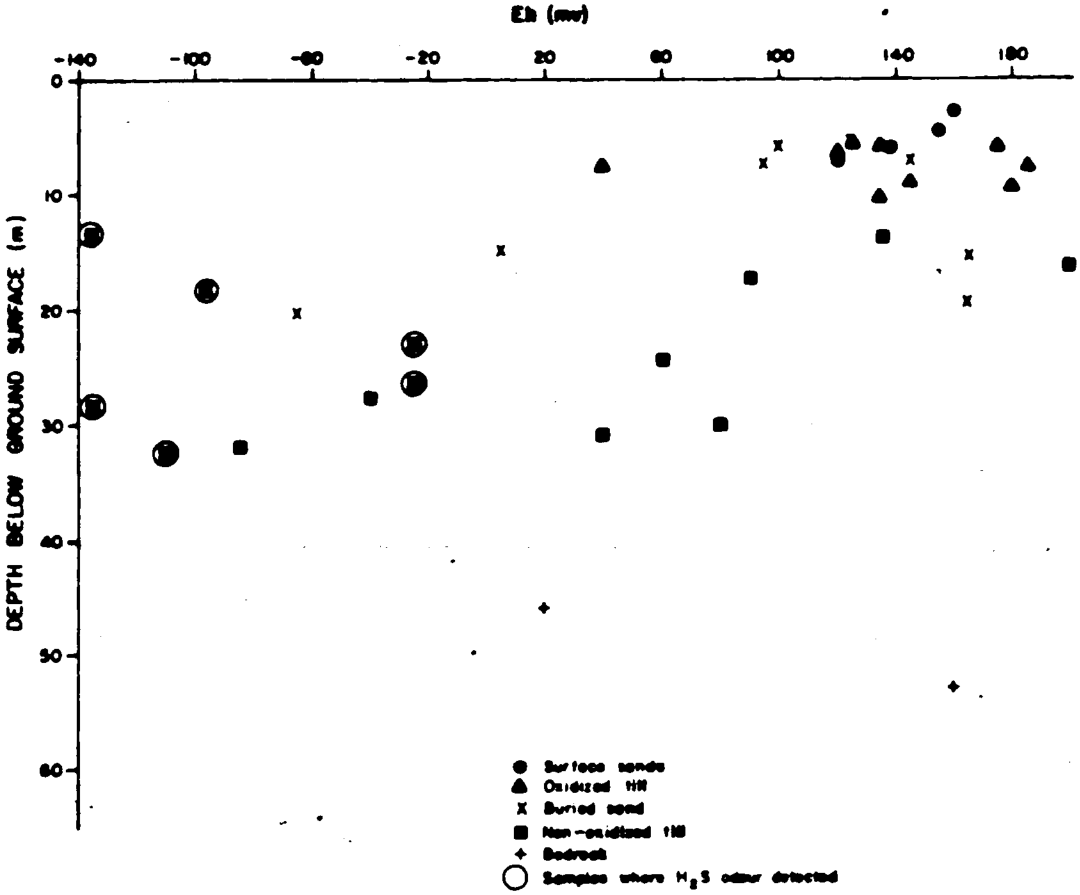


FIGURE 16: Field Eh versus Depth from which Water Sample was taken (1st Sample Set).

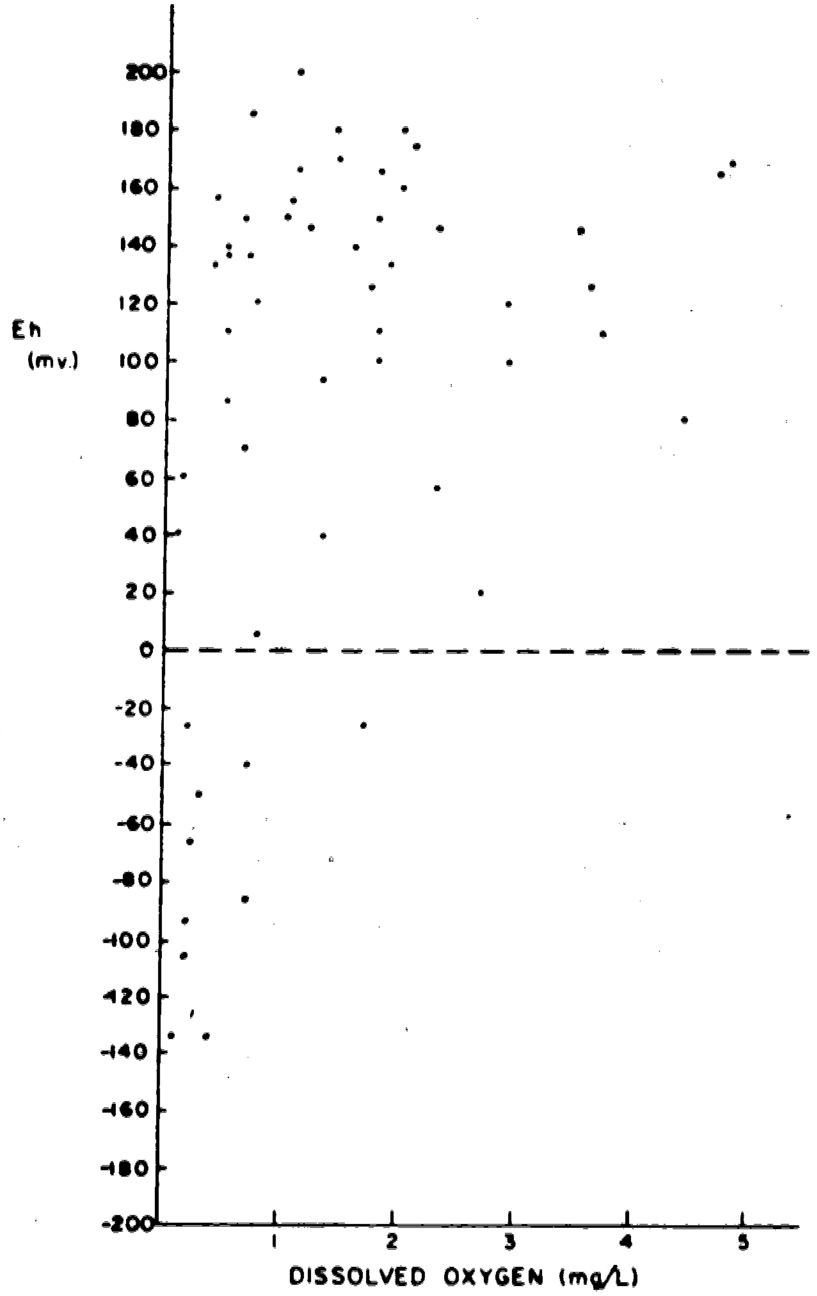


FIGURE 17: Field Eh of Water Sample versus Dissolved Oxygen Concentration (1st Sample Set).

possible. The implications of this with respect to a hydrochemical evolution model will be discussed later.

4.5 Chemistry of Samples from the Surface Sands

Two test sites, 1814 and 1818, have piezometers completed in the surface sands; these range from 2.8m to 7.1m beneath ground surface. Site 1818 is in the northern portion of the study area while 1814 is on the eastern side (figure 3). The water table well at 1818 is also completed in sands therefore data is included from this hole. The results of field measurements and laboratory analyses performed on water samples from the surface sands have already been presented in tables 4 to 6. The increasing concentration of many of the ions with increasing depth is obvious.

Tables 4 to 6 also show that water samples from the surface sands at site 1814 have much greater ionic concentrations than water samples from the surface sands at site 1818. A homogeneity of variance test (F-test), performed on the total dissolved solids variances from the two sites showed that the samples could not be assumed to have been derived from the same population and so no difference of means t-test could be performed on the data. In order to illustrate the differences between the two sites, mean ionic concentrations and sample standard deviations (s.d.) have been computed using data from all three sample sets, for the surface sands at sites 1814 and 1818 separately as well as for the combined data from the two sites. The results of these computations as well as the ranges of major ionic constituents are given in table 8.

In some cases nitrate-nitrogen and chloride concentrations in the water samples were below laboratory detection limits, that is, less than 0.1mg/L. Concentrations of these ions were therefore assumed to be zero during computation of means and standard deviations.

Table 8 shows that the major ions present in the water samples from the surface sands at 1814 are sodium and sulphate. However these are also the two species which have major differences in concentration between sites 1814 and 1818. At site 1818 the bicarbonate ion is dominant. The differences between water samples from similar deposits at different test sites will be considered in greater detail during the discussion.

4.6 Chemistry of Samples from the Oxidized Till

The oxidized till is much more extensive across the study site than the surface sand deposits (figures 5 to 7). For this reason there are more piezometers completed in oxidized till than in the surface sands. The intake zones of these oxidized till piezometers range from 6.0m to 10.6m beneath ground surface. At all piezometer nests where piezometers are completed in oxidized till, with the

TABLE 8

Summary Chemistry of Groundwater Samples from the Surface Sands

	Surface Sands at 1814 and 1818			Surface Sands at 1814			Surface Sands at 1818		
	Range	Mean	s.d.	Range	Mean	s.d.	Range	Mean	s.d.
Field pH	7.5 to 7.9	7.62	0.15	7.5 to 7.9	7.67	0.20	7.5 to 7.7	7.66	0.07
Lab pH	7.7 to 8.2	7.97	0.19	7.9 to 8.3	8.06	0.16	7.7 to 8.2	7.9	0.20
Field EC (mhos)	0.38 to 9.1	3.06	3.50	3.35 to 9.1	6.55	2.52	0.38 to 0.49	0.44	0.04
Lab EC (mhos)	0.55 to 12.01	4.1	4.75	4.53 to 12.01	8.76	3.55	0.55 to 0.60	0.59	0.02
TDS (mg/L)	488.9 to 11565.4	3788.5	4558.4	3827.3 to 11565.4	8149.4	3752.6	488.9 to 545.1	517.9	15.80
Calcium (mg/L)	46.1 to 440.9	182.0	144.2	138.3 to 440.9	300.6	116.8	46.1 to 62.1	58.1	5.95
Magnesium (mg/L)	19.4 to 498.4	161.0	186.5	189.6 to 498.4	344.4	140.4	19.4 to 32.8	23.4	5.80
Sodium (mg/L)	29.9 to 2597.8	761.1	1020.2	747.2 to 2597.8	1723.1	873.5	29.9 to 43.7	39.7	5.4
Potassium (mg/L)	2.3 to 24.6	8.7	8.4	7.4 to 24.6	16.2	8.3	2.3 to 4.7	3.2	0.45
Chloride (mg/L)	<0.1 to 255.2	63.3	95.1	39.0 to 255.2	142.4	102.4	<0.1 to 7.1	4.0	2.9
Sulphate (mg/L)	28.8 to 7348.7	2223.4	2997.9	2180.6 to 7348.7	5065.6	2482.7	28.8 to 81.6	56.7	15.3
Bicarbonate (mg/L)	296.5 to 602.8	424.1	120.1	512.5 to 602.8	554.5	39.1	296.5 to 341.7	326.3	13.7
Nitrate-Nitrogen (mg/L)	<0.1 to 5.4	1.1	1.75	0.1 to 5.4	2.48	2.0	<0.1 to 0.3	0.08	0.11
Eh (mv)	+120 to +155	+140	15.5						

exception of sites 1814 and 1818, the till directly underlies the soil horizons. At sites 1814 and 1818 between 5m and 8m of surface sands overlie the oxidized till (figures 5 to 7).

Tables 4 to 6 present the results of field measurements and laboratory analyses performed on water samples taken from piezometers completed in the oxidized till. The sample from site 1818 has considerably lower concentrations of all ions except nitrate-nitrogen, than samples from the other sites.

Table 9 gives mean ionic concentrations, sample standard deviations and ranges of major ionic constituents in water samples taken from the oxidized till. Since concentrations in the sample from site 1818 are so much lower than those from other sites, means have been computed for each species first including results from site 1818 and then excluding results from this site. Table 9 clearly shows the depression of means and increase in sample standard deviation which results from the inclusion of data from site 1818.

Sodium and sulphate are present in the highest ionic concentrations in the water samples taken from the oxidized till. Nitrate-nitrogen concentration is very variable, and with the exception of sites 1817 and 1820 in the central region of the field area, nitrate-nitrogen occurs at only very low concentrations. Comparison of tables 8 and 9 illustrates that concentrations of all major ion species are

TABLE 9

Summary Chemistry of Groundwater Samples from the Oxidized Till

	Including Data from 1818			Excluding Data from 1818		
	Range	Mean	s.d.	Range	Mean	s.d.
Field pH	6.4 to 7.7	7.15	0.38	6.4 to 7.7	7.08	0.34
Lab pH	7.0 to 8.2	7.53	0.32	7.0 to 8.0	7.47	0.28
Field EC (mhos)	0.47 to 9.5	6.14	2.51	5.25 to 9.5	7.00	1.36
Lab EC (mhos)	0.70 to 12.86	8.84	3.68	7.72 to 12.86	10.06	1.88
TDS (mg/L)	609.2 to 12940.6	7793.4	3223.2	6637.0 to 12940.6	9063.9	1824.5
Calcium (mg/L)	40.1 to 543.1	391.5	140.8	348.7 to 543.1	441.6	41.2
Magnesium (mg/L)	9.7 to 459.5	257.5	109.7	203.0 to 459.5	294.1	59.5
Sodium (mg/L)	108.0 to 2942.5	1726.8	812.6	1220.7 to 2942.5	1957.8	557.7
Potassium (mg/L)	4.7 to 35.6	26.2	9.03	24.2 to 35.6	29.3	3.8
Chloride (mg/L)	<0.1 to 251.0	161.9	104.2	56.7 to 251.0	184.6	90.4
Sulphate (mg/L)	72.0 to 7108.5	4585.9	2058.1	3467.8 to 7108.5	5183.2	1192.2
Bicarbonate (mg/L)	361.2 to 1449.6	825.2	278.6	688.9 to 1449.6	891.3	229.5
Nitrate-Nitrogen (mg/L)	<0.1 to 130.0	36.6	54.9	<0.1 to 130.0	46.5	57.0
Eh (mv)	+40 to +200	+140	49.0			

greater in the oxidized till water samples than in those from the surface sands. Further discussion of this observation is presented later.

4.7 Chemistry of Samples from the Buried Sands

Buried sand lenses are common beneath the field area, however because of the laterally discontinuous nature of many lenses, piezometer intake zones could not be accurately positioned within them. The water samples from the buried sands which are described in this section come from larger sand lenses which occur beneath piezometer nests 1813, 1814 and 1817. Sites 1813 and 1814 are on the east side of the field area while 1817 is in the centre (figure 3).

The sand lens beneath sites 1813 and 1814 shown in the cross sections (figures 5 and 6) is continuous between the two sites and represents a buried channel deposit which presently drains toward the south; it is described earlier in this thesis and in the TID Return Flow Study (Alberta Agriculture, 1981). The four piezometers in this sand unit range in depth from 14.8m to 20.3m below ground surface. The sands underlie a total of 12m to 15m of surface sands and oxidized till and are approximately 8m thick beneath each testsite.

The three other piezometers in buried sands are located at piezometer nest site 1817. Here the sands are laterally discontinuous, lie beneath 5m to 6m of oxidized till, and are 3m to 4m thick. Piezometers in this sand lens are located between 5.7m and 7.4m below ground surface.

Table 10 summarizes the chemical analysis results for water samples from the buried sands. The original data is given in tables 4 to 6. Table 10 gives the range, mean and sample standard deviation for each of the major ion species present in the samples from the buried sands. The previously mentioned relationship between laboratory and field pH and EC are evident in this table. The main feature of the water samples from these buried sand deposits is the highly variable ionic concentration of major species. This is reflected in the total dissolved solid concentrations which range from 841mg/L to 8833.6mg/L. Once again sodium and sulphate ions are the major contributors to the total dissolved solids.

Examination of the original data given in tables 4 to 6 shows that concentrations of ionic species at site 1813 and 1814 are generally lower than at site 1817. Ionic concentrations increase with depth through the sands beneath sites 1813 and 1814. Total dissolved solids in the upper piezometer in the sand lens at site 1813 decreased markedly through the field season. Total dissolved solids do not increase with depth in the sand lens beneath site 1817, instead, they decrease in the upper meter of the sand lens and increase again at its base.

Hydraulic head measurement taken during 1979 and 1980 (Alberta Agriculture, 1981), indicate a downward hydraulic

TABLE 10

Summary Chemistry of Groundwater Samples from the Buried Sands

	Range	Mean	s.d.
Field pH	6.8 to 7.4	7.13	0.18
Lab pH	7.2 to 7.9	7.67	0.20
Field EC (mhos)	0.7 to 8.25	4.78	2.23
Lab EC (mhos)	0.96 to 11.12	6.93	3.14
TDS (mg/L)	841.0 to 8813.6	5718.9	2350.7
Calcium (mg/L)	106.2 to 629.3	433.0	149.9
Magnesium (mg/L)	36.5 to 328.2	208.8	81.1
Sodium (mg/L)	55.2 to 1694.4	1137.7	670.1
Potassium (mg/L)	4.7 to 30.5	19.6	7.2
Chloride (mg/L)	3.5 to 290.7	123.9	88.7
Sulphate (mg/L)	326.6 to 4447.6	3106.9	1305.0
Bicarbonate (mg/L)	232.0 to 873.8	507.2	228.5
Nitrate-Nitrogen (mg/L)	<0.1 to 576	172.8	207.7
Eh (mv)	-65 to +165	+92	98.5

gradient through the buried sands at sites 1813 and 1814. Hydraulic gradients also indicate lateral flow from site 1814 to 1813. At site 1817, hydraulic gradients through the buried sands are of variable direction and magnitude throughout the field season.

Comparison of tables 8 and 10 illustrates the difference in chemical quality between water samples from surface sands and those from buried sands. Concentrations of all major ion species are greater in the water samples taken from the buried sands than they are in samples from the surface sands. Comparison of tables 9 and 10 shows that mean ionic concentrations of all species except calcium and nitrate-nitrogen are not as great in the water from the buried sands as concentrations in water samples taken from the oxidized till which overlie the sands. With respect to this point one should note that the mean ionic concentrations given for the oxidized till in table 9 include water samples from greater depths than the top of the buried sand deposits.

Changes in the chemical quality of the groundwater through the buried sands and the relationships between water quality in the buried sands and surrounding tills will be examined in the discussion.

4.8 Chemistry of Samples from the Non-oxidized Till

The top of the non-oxidized till occurs at depths between 10m and 25m beneath most of the study area, its upper surface is deepest where it underlies the major sand lens beneath piezometer nests 1813 and 1814 (figures 5 and 6). Non-oxidized till extends to depths of 35m to 40m where it overlies the Saskatchewan sands and gravels. Piezometers completed in this unit are spread across the study site at depths ranging from 13.5m to 32.5m below ground surface. Fewer of these piezometers were sampled in the second and third data sets than in the first because of their slow response to bailing.

Results of field measurements and chemical analyses for major ionic constituents in water samples taken from the non-oxidized till unit have been presented in tables 4 to 6. A similar pattern to that shown by water samples from the oxidized till is evident, in that samples from the non-oxidized till at sites 1811 and 1818 have lower concentrations of all major ion species except nitrate-nitrogen than samples taken from the non-oxidized till at other sites across the field area.

Table 11 is a summary of the data given in tables 4 to 6 and gives the mean, sample standard deviation and range of each variable, both including and excluding data from sites 1811 and 1818. This table shows that even the deepest water samples from the field area have a moderate concentration range for all major ionic species.

TABLE 11

Summary Chemistry of Groundwater Samples from the Non-Oxidized Till

	Including Data from 1811 & 1818			Excluding Data from 1811 & 1818		
	Range	Mean	s.d.	Range	Mean	s.d.
Field pH	6.7 to 8.2	7.24	0.39	6.7 to 7.5	7.0	0.25
Lab pH	6.9 to 8.2	7.56	0.29	6.9 to 7.8	7.4	0.26
Field EC (mhos)	0.65 to 5.0	2.50	1.58	2.9 to 5.0	3.75	0.66
Lab EC (mhos)	0.93 to 7.2	3.66	2.36	4.3 to 7.2	5.53	0.95
TDS (mg/L)	657.5 to 6386.6	3203.4	2197.2	3949.3 to 6386.6	4992.0	849.9
Calcium (mg/L)	7.4 to 460.9	237.2	174.2	240.5 to 460.9	374.5	68.6
Magnesium (mg/L)	7.3 to 226.1	98.2	76.7	100.9 to 226.1	157.4	36.4
Sodium (mg/L)	91.9 to 1333.4	581.7	413.0	664.4 to 1333.4	881.2	255.7
Potassium (mg/L)	5.5 to 21.1	13.9	5.14	14.1 to 21.1	17.4	2.4
Chloride (mg/L)	3.5 to 60.3	26.8	20.5	14.2 to 60.3	40.6	15.4
Sulphate (mg/L)	82.1 to 3799.2	1582.8	1315.6	1926.0 to 3799.2	2598.4	628.3
Bicarbonate (mg/L)	395.4 to 1236.0	709.8	294.5	538.2 to 1236.0	887.1	255.8
Nitrate-Nitrogen (mg/L)	<0.1 to 26.4	2.3	6.3	<0.1 to 26.4	4.6	9.4
Eh (mv)	+200 to -135	+3.8	106			

Sulphate, bicarbonate and sodium are present in the highest ionic concentrations in water samples from the non-oxidized till, although the sodium and sulphate concentrations are not as high as in the other units. Comparison of tables 9 and 11 illustrates the differences between water samples from the two types of till. Total dissolved solid concentrations are considerably lower in water samples from the non-oxidized unit than in samples from the oxidized till. Although most major ionic species in the non-oxidized till water samples do have lower concentrations, it is the difference in sodium and sulphate ion concentrations between samples from the two units which are the main contributors to this difference in total dissolved solids.

In contrast with the trends in most of the ionic species in water samples from the two till units, mean bicarbonate ion concentrations are almost equal. Even though the bicarbonate ion has a slightly higher mean concentration in samples from the oxidized till (891.3mg/L), than in samples from the non-oxidized till (887.1mg/L), it contributes a larger proportion of the total dissolved solids concentration in the non-oxidized till, (approximately 18%) than it does in samples from the oxidized till (approximately 10%).

The chemical nature of the groundwater within the non-oxidized till unit and the reasons for the differences in water chemistry between the two till units will be examined during the discussion.

A summary of the ranges and means of the total dissolved solid concentrations in water samples from the four units described above is given in figure 18. Figure 19 is a plot of mean concentration of each major ion species through a generalized cross-section and shows the general trends which occur through the profile. The means used here include data from sites 1811 and 1818. The depths at which the mean concentrations are plotted in each of the four geologic units are the mean depths from which all samples collected from that unit across the field area were taken. Changes between types of deposit will be examined in greater detail during the discussion.

4.9 Results obtained from WATEQF

The WATEQF computer programme not only calculates total ionic concentrations of major species present in a water sample, but also gives a number of other data which are useful when attempting to develop a hydrochemical evolution model for a groundwater flow system. This section will summarize some of the more important output variables obtained once the water sample laboratory results had been processed by WATEQF.

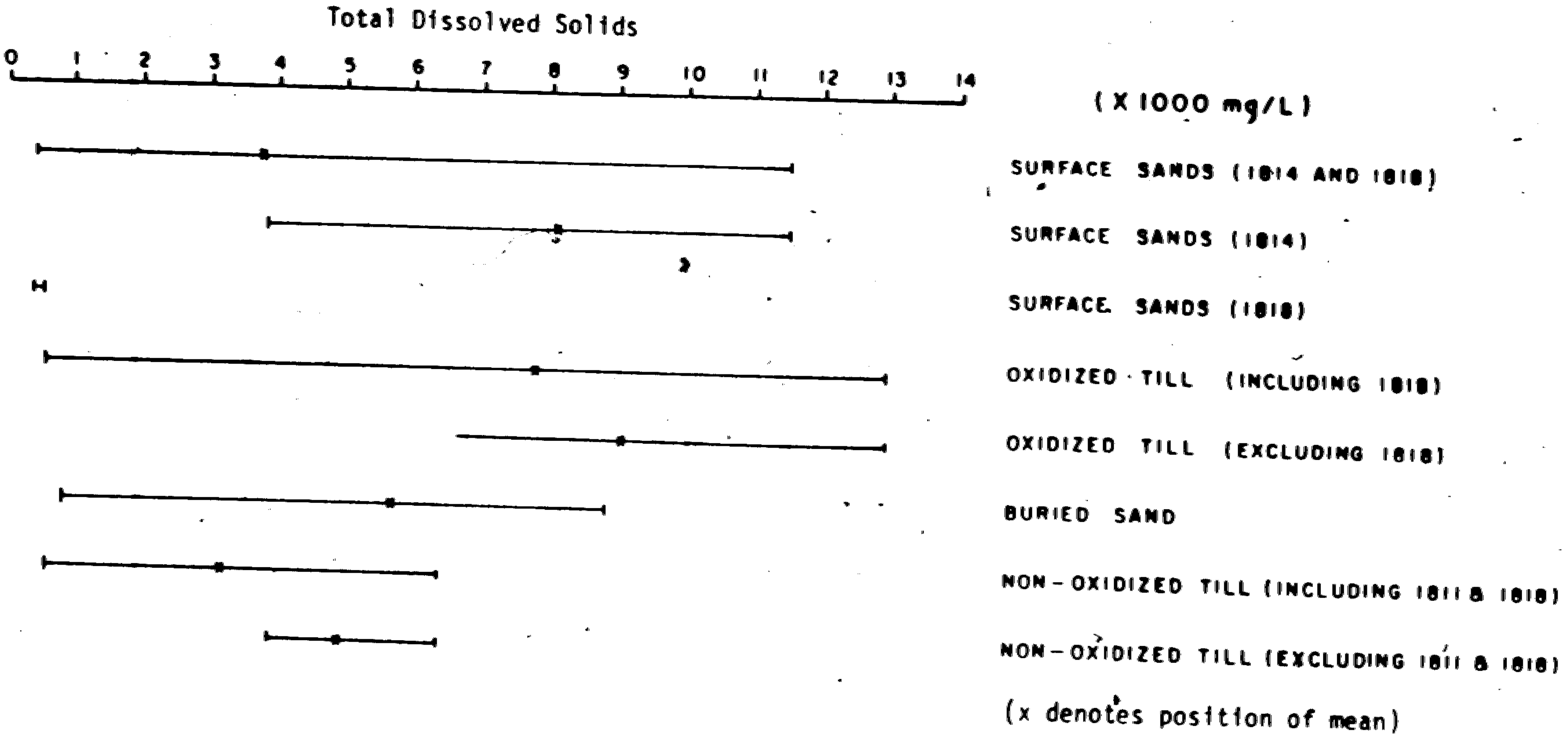


FIGURE 18: Means and Ranges of Total Dissolved Solids Concentrations in Water Samples from Different Overburden Units and Piezometer Nests.

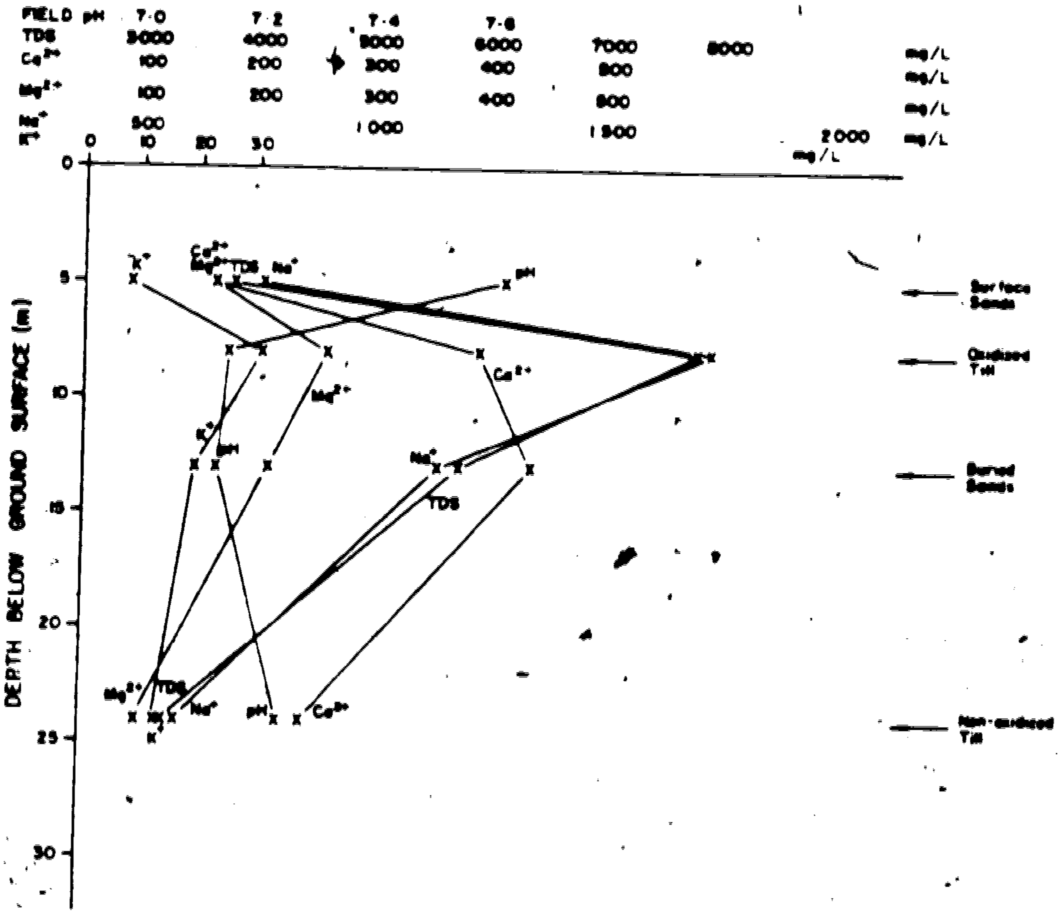


FIGURE 19a: Changes in Mean Cation Concentrations in the Groundwater Through a Generalized Profile

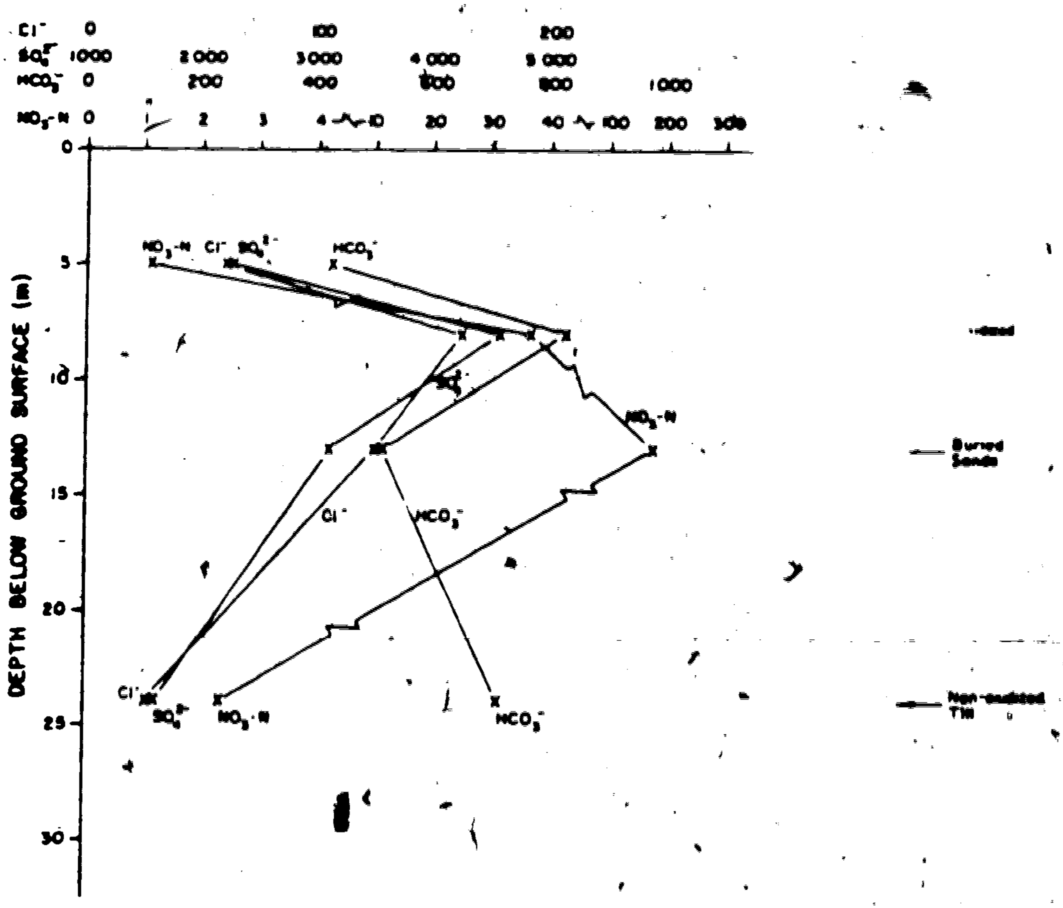


FIGURE 19b: Changes in Mean Anion Concentrations in the Groundwater Through a Generalized Profile.

4.9.1 Saturation Indices

Some of the most useful variables obtained from the WATEQF programme are the saturation indices of the water samples with respect to a number of mineral species. The saturation index, (SI), of a water sample is a measure of the thermodynamic state of the solution relative to equilibrium with a specific solid phase mineral (Wigley, 1977). The equation which defines the saturation index is;

$$SI=\log(IAP/K(eq)) \qquad (4)$$

where IAP is the ion activity product and K(eq) is the equilibrium constant for the dissolution reaction of the species in question. An example can be used to illustrate this point. The simplified equation for the solution of calcite is;

$$CaCO_3(s) \longrightarrow Ca^{2+} + CO_3^{2-} \qquad (5)$$

where (s) denotes the solid phase. For this reaction to be at equilibrium, the products of the activities of the two ionic species on the right hand side of equation 5, that is the ion activity product, must equal the equilibrium constant for this reaction, where,

$$aCa^{2+}.aCO_3^{2-} = K(eq) \qquad (6)$$

at equlibrium. If the ion activity product is less than K(eq) then dissolution will occur in the attempt to attain equilibrium, conversely if the ion activity product is greater than K(eq) then precipitation will occur.

Returning to equation 4 for the saturation index, and following the reasoning given above, if the saturation index is equal to zero, (that is IAP=K(eq)), then the dissolution reaction for the mineral species in question is at equilibrium and the sample is saturated with respect to that species. When the ion activity product is less than K(eq), the saturation index will be negative, hence this indicates undersaturation with respect to the mineral species being considered and conversely a positive saturation index indicates supersaturation with respect to that species.

Table 12 summarizes the saturation indices obtained from WATEQF output with respect to the mineral species which are most often saturated in the water samples, that is calcite, dolomite and gypsum. Only results from the first two data sets are presented. For the third data set only, laboratory pH was availible and a sensitivity test on one set of data, running WATEQF first with field pH and then with laboratory pH , revealed that saturation indices were significantly influenced by the differences in field and laboratory pH; therefore saturation indices from the third data set could not be used. The results are once again subdivided according to geologic unit from which the water sample was taken. The saturation indices are given with respect to the mineral species, calcite, dolomite and gypsum.

Saturation indices of samples taken from the surface sands (table 12), show that all samples are supersaturated with respect to calcite while the shallow samples are supersaturated with respect to dolomite. This is probably

TABLE 12

Saturation Indices of Water Samples With Respect to Calcite, Dolomite and Gypsum

Depth (m)	Hole	Calcite	Dolomite	Gypsum
Surface Sand				
2.8	1814-2.8	0.6/0.7	1.5/1.5	-0.6/-0.3
4.4	1814-4.4	0.5/0.4	1.2/1.0	0.0/0.0
5.9	1818-6	0.2/0.1	0.0/-0.2	-1.9/-1.8
6.1	1818-7(WTW)	---/0.2	---/0.3	---/-2.0
7.1	1818-5	0.2/0.0	-0.1/-0.3	-1.8/-1.8
Oxidized Till				
6.0	1820-N(WTW)	0.5/0.3	1.0/0.7	0.1/0.0
6.1	1816-5(WTW)	0.3/0.2	0.6/0.4	0.1/0.0
6.9	1814-6.9	0.6/0.4	1.2/0.9	0.0/0.0
7.5	1820-S	0.2/0.2	0.4/0.2	0.1/0.1
7.6	1816-4	0.5/---	0.6/---	0.1/---
9.2	1814-5	0.3/0.0	0.3/-0.1	0.1/0.0
9.4	1812-5	0.2/-0.3	0.2/-0.8	0.0/0.0
10.6	1818-4	0.1/0.0	-0.3/-0.4	-1.9/-1.8
Buried Sand				
5.7	1817-5.7	0.3/0.2	0.6/0.3	0.1/0.1
7.0	1817-7	0.2/0.2	0.1/0.2	0.1/0.1
7.4	1817-7.4	0.3/0.1	0.4/0.1	0.2/0.2
14.8	1814-4	0.1/0.0	-0.1/-0.2	-0.2/-0.3
15.4	1813-5	0.4/0.2	0.5/0.0	-0.1/0.7
19.5	1813-4	0.4/0.1	0.4/-0.1	0.0/0.0
20.3	1814-3	0.2/0.1	0.1/0.0	-0.1/-0.1
Non-oxidized Till				
13.5	1816-3	0.1/0.1	-0.2/-0.1	0.0/0.0
13.8	1818-3	0.0/0.0	-0.6/0.5	-1.7/-1.7
16.4	1817-6	0.2/0.2	0.3/0.3	0.0/0.0
17.4	1811-3	0.1/-0.1	0.0/-0.7	-1.5/-1.8
22.1	1811-2	0.4/0.0	0.4/-0.3	-1.6/-1.6
22.9	1812-3	0.5/0.1	0.7/0.0	-0.3/-0.3
24.5	1811-1	0.3/0.0	0.2/-0.3	-2.0/-2.1
26.7	1817-5	0.1/0.2	0.0/0.2	0.0/0.0
27.7	1818-2	0.1/---	-0.3/---	-1.6/---
28.6	1816-2	0.5/---	0.6/---	-0.2/---
30.2	1813-3	0.7/0.4	1.2/0.6	-0.1/0.1
32.0	1814-2	0.4/0.3	0.3/0.4	-0.2/-0.3
32.5	1817-4	0.4/0.2	0.5/0.2	-0.1/0.1

NB: Results from first data set precede results from second data set.

the result of concentration by evaporation and evapotranspiration in the unsaturated zone. Only the sample from 4.4 meters at site 1814 is saturated with respect to gypsum, the rest are undersaturated.

Saturation indices for samples from the oxidized tills are also given in table 12. Most samples are saturated or supersaturated with respect to calcite. Samples are supersaturated with respect to dolomite in data set 1 with the exception of the sample from 10.6m at site 1818. In data set 2 most of the deeper samples are undersaturated with respect to dolomite. Shallow samples are saturated with respect to gypsum while the deepest sample is undersaturated.

In the water samples from the buried sands supersaturation with respect to calcite is common to all but one of the samples which is saturated (SI=0.0, table 12). Dolomite and gypsum saturation indices are variable with depth however a number of the samples from the buried sand deposits are saturated or supersaturated with respect to these two minerals.

Water samples from the non-oxidized till include the deepest samples from the field area. The saturation indices of these samples are also given in table 12. All but one of the samples are saturated or supersaturated with respect to calcite. The degree of saturation with respect to dolomite varies with depth and between the two data sets. The majority of samples are undersaturated with respect to gypsum.

4.9.2 Partial Pressure of Carbon Dioxide (pCO_2)

Another useful variable computed by **WATEQF** for each of the water samples is the partial pressure of carbon dioxide (pCO_2atm) with which the carbon dioxide dissolved in the water sample is in equilibrium. In the atmosphere the partial pressure of carbon dioxide is 3.16×10^{-4}atm (or $1 \times 10^{-3.5}$atm). pCO_2 values calculated by **WATEQF** are presented in table 13 where they are subdivided according to geologic unit from which the water sample was taken. All pCO_2 values are greater than atmospheric pCO_2, indicating that carbon dioxide was dissolved into the water in a carbon dioxide rich environment. This is caused by a number of processes occuring in the deposits; these will be discussed later.

The effect of using laboratory pH instead of field pH in the computations for data set 3 can be clearly seen in the lower pCO_2 values which result. One of the predominant influences on the pH of groundwater is the amount of carbon dioxide dissolved in the water. Increasing the amount of carbon dioxide in solution increases the acidity of the groundwater and decreases its pH. **WATEQF** is designed to calculate pCO_2 for the water samples by taking account of *in situ* temperature and pH among other variables. Therefore, the higher the input pH the lower the pCO_2 value in the output. Since laboratory pH is usually greater than field pH, pCO_2 is consistently lower for samples in data set 3

TABLE 13

Partial Pressure of Carbon Dioxide (pCO_2 atm) With Which the Groundwater Samples Are in Equilibrium

Depth (m)	Hole	Data Set 1	Data Set 2	Data Set 3
Surface Sand				
2.8	1814-2.8	4.5×10^{-3}	5.4×10^{-3}	2.7×10^{-3}
4.4	1814-4.4	1.1×10^{-2}	1.1×10^{-2}	4.6×10^{-3}
5.9	1818-6	5.6×10^{-3}	8.1×10^{-3}	4.2×10^{-3}
6.1	1818-7(WTW)	---	5.6×10^{-3}	3.4×10^{-3}
7.1	1818-5	6.3×10^{-3}	8.5×10^{-3}	4.7×10^{-3}
Oxidized Till				
6.0	1820-N(WTW)	1.75×10^{-2}	2.7×10^{-2}	---
6.1	1816-5(WTW)	2.6×10^{-2}	2.6×10^{-2}	8.5×10^{-3}
6.9	1814-6.9	1.5×10^{-2}	1.9×10^{-2}	8.9×10^{-3}
7.5	1820-5	4.5×10^{-2}	5.8×10^{-2}	---
7.6	1816-4	3.5×10^{-2}	---	---
9.2	1814-5	6.9×10^{-2}	1.2×10^{-1}	5.2×10^{-2}
9.4	1812-5	1.5×10^{-1}	2.45×10^{-1}	6.0×10^{-2}
10.6	1818-4	5.6×10^{-3}	7.1×10^{-3}	3.5×10^{-3}
Buried Sand				
5.7	1817-5.7	1.5×10^{-2}	1.1×10^{-2}	4.8×10^{-3}
7.0	1817-7	2.2×10^{-2}	1.5×10^{-2}	5.7×10^{-3}
7.4	1817-7.4	9.9×10^{-3}	1.8×10^{-2}	6.1×10^{-3}
14.8	1814-4	6.4×10^{-2}	8.5×10^{-2}	3.8×10^{-2}
15.4	1813-5	1.0×10^{-2}	9.8×10^{-3}	2.7×10^{-3}
19.5	1813-4	1.8×10^{-2}	2.5×10^{-2}	8.7×10^{-3}
20.3	1814-3	5.4×10^{-2}	6.9×10^{-2}	1.1×10^{-2}
Non-oxidized Till				
13.5	1816-3	8.5×10^{-2}	8.3×10^{-2}	3.5×10^{-2}
13.8	1818-3	9.35×10^{-3}	9.4×10^{-3}	4.8×10^{-3}
16.4	1817-6	1.3×10^{-1}	1.2×10^{-1}	3.8×10^{-2}
17.4	1811-3	1.1×10^{-2}	1.8×10^{-2}	6.2×10^{-3}
22.1	1811-2	5.1×10^{-3}	1.2×10^{-2}	6.4×10^{-3}
22.9	1812-3	6.2×10^{-2}	1.1×10^{-1}	2.7×10^{-2}
24.5	1811-1	3.6×10^{-3}	4.9×10^{-3}	4.1×10^{-3}
26.7	1817-5	1.7×10^{-1}	1.6×10^{-1}	---
27.7	1818-2	1.5×10^{-2}	---	---
28.6	1816-2	2.3×10^{-2}	---	---
30.2	1813-3	1.25×10^{-2}	2.0×10^{-2}	7.8×10^{-3}
32.0	1814-2	3.5×10^{-2}	4.3×10^{-2}	---
32.5	1817-4	1.1×10^{-1}	1.5×10^{-1}	---

than for the other data sets.

Water samples from the surface sands have the lowest mean pCO_2 value (6.1×10^{-3}atm, s.d.=2.6×10^{-3}atm), pCO_2 then increases into the oxidized tills which have the highest mean value in the profile (5.14×10^{-2}atm, s.d.=5.8×10^{-2}atm) as well as the widest range in pCO_2. pCO_2 concentrations in the groundwater decrease into the buried sands although concentrations are still higher than in the surface sands and then increase again into the non-oxidized till. Non-oxidized till and buried sand water samples have mean carbon dioxide partial pressures of 4.12×10^{-2}atm (s.d.=4.23×10^{-2}atm) and 2.39×10^{-2}atm (s.d.=2.38×10^{-2}atm) respectively.

4.9.3 Ca:Mg Molar ratios

WATEQF computes molar ratios of various pairs of ions in the water samples from total ion concentrations. One of the most useful of these is the Ca:Mg molar ratio. Assuming the major source of calcium and magnesium ions in the groundwater system is the dissolution of calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$), and that no process occurs, such as precipitation of calcite, dolomite or gypsum which might alter the concentrations of these ions, Ca:Mg molar ratios greater than unity would be expected since both calcite and dolomite contribute calcium to the groundwater whereas only dolomite contributes magnesium. Table 14 presents the Ca:Mg molar ratios obtained from WATEQF; some of the samples have

TABLE 14

Ca:Mg Molar Ratios in Groundwater Samples

Depth (m)	Hole	Data Set 1	Data Set 2	Data Set 3
Surface Sand				
2.8	1814-2.8	0.44	0.56	0.66
4.4	1814-4.4	0.49	0.48	0.58
5.9	1818-6	0.71	1.82	1.76
6.1	1818-7(WTW)	----	0.96	0.85
7.1	1818-5	1.82	1.94	1.94
Oxidized Till				
6.0	1820-N(WTW)	0.79	0.80	0.82
6.1	1816-5(WTW)	0.81	0.85	0.90
6.9	1814-6.9	0.57	0.64	0.76
7.5	1820-S	1.01	0.96	1.00
7.6	1816-4	1.62	----	----
9.2	1814-5	1.09	1.02	1.07
9.4	1812-5	1.15	1.04	1.07
10.6	1818-4	2.50	2.33	2.22
Buried Sand				
5.7	1817-5.7	1.01	1.15	1.23
7.0	1817-7	1.20	1.15	1.18
7.4	1817-7.4	1.16	1.33	1.32
14.8	1814-4	1.62	0.97	1.45
15.4	1813-5	1.84	1.85	1.77
19.5	1813-4	1.75	1.41	1.53
20.3	1814-3	1.36	0.97	0.72
Non-oxidized Till				
13.5	1816-3	1.72	1.91	1.85
13.8	1818-3	2.83	2.57	1.85
16.4	1817-6	1.01	1.04	0.97
17.4	1811-3	1.73	2.00	2.33
22.1	1811-2	1.78	1.80	2.05
22.9	1812-3	1.61	0.83	1.55
24.5	1811-1	2.00	1.75	1.60
26.7	1817-5	1.33	1.11	----
27.7	1818-2	2.37	----	----
28.6	1816-2	2.02	----	----
30.2	1813-3	1.49	1.13	1.50
32.0	1814 2	1.67	0.96	----
32.5	1817-4	1.53	1.18	----

Ca:Mg molar ratios less than unity. This must be the result of the relative concentration of Mg^{2+} ions with respect to Ca^{2+} ions in the water samples. Evaporation and evapotranspiration can produce this effect since they concentrate ionic species in the water. This causes calcium salts to precipitate before magnesium salts because of their lower solubility and results in the relative enrichment of magnesium.

Examination of table 14 seems to support the idea of evaporation and evapotranspiration from the unsaturated zone above the water table leading to lower Ca:Mg molar ratios since, in most cases, it is the shallow samples which have Ca:Mg molar ratios less than unity. However, some deeper samples also show these trends. For example, the sample at 20.3m in the buried sands has a Ca:Mg molar ratio of less than one in two of the three data sets. To account for this some other process leading to the relative enrichment of Mg^{2+} must be invoked. This will be investigated in greater detail in the discussion.

Ca:Mg molar ratios in table 14 are divided according to geologic unit from which the water samples were taken and mean Ca:Mg ratio for each unit was computed. Mean Ca:Mg molar ratio in the surface sands is 0.86 (s.d.=0.65) and in the oxidized till it is greater than one at 1.19 (s.d.=0.61). Ca:Mg molar ratio continues to increase through the buried sands and into the non-oxidized till from a mean of 1.42 (s.d.=0.32) in the buried sand to 1.78 (s.d.=0.46) in the non-oxidized till.

4.10 Results of 5:1 Aqueous Extract Analyses

The procedure for obtaining the 5:1 aqueous extracts from samples of deposits beneath the study site has been outlined in the methods section. The results of the chemical analyses of these 5:1 extracts are presented in table 15; concentrations of all ionic species are given in meq/L and site locations can be found on figure 3. Samples for which extracts were analysed include seven from surface sands, eleven from oxidized tills and one from the buried sands. Unfortunately, no samples were taken from the non-oxidized tills however, these would have been useful and should be collected during any further research.

Graphical plots of each major ion species present in the 5:1 extract versus depth from which the till or sand sample was taken are presented in figures 20(a) to 20(j). Some correlation between depth and concentration of ions within the extracts was expected; however this was not the case for any of the ions. Electrical conductivity (EC), (figure 20(a)) which is a direct measure of the total dissolved solids concentrations in the extracts also shows no correlation with depth

A marked differentiation between extracts from the till and the sands can be seen in figure 20(a) showing that the electrical conductivities of the extracts from the till

TABLE 15

Results of 5:1 Aqueous Extract Analyses (meq/L)

Unit	Depth of Sample	Site Number	Extract Sample Number	pH	$Ca^{2+} + Mg^{2+}$	Na^+	K^+	Cl^-	SO_4^{2-}	$CO_3^- - HCO_3^-$	NO_3^-	NH_4^+	EC mhos
Surface Sand	0-0.3 m	1818	1	7.4	6.0	1.0	0.20	0.2	2.65	1.94	0.98	0.17	0.70
Surface Sand	0.6 m	4417	2	8.7	6.5	2.0	0.12	0.2	2.25	1.62	2.75	0.28	0.78
Surface Sand	0.75 m	WTW#1	3	8.7	2.25	5.1	0.13	0.5	3.35	1.66	0.39	0.42	0.74
Surface Sand	2.25 m	1814	4	8.6	8.0	5.8	0.19	0.13	10.5	1.07	0.11	0.11	1.26
Surface Sand	3.0 m	1814	5	8.8	2.5	6.2	0.10	0.13	4.55	1.27	0.12	0.11	0.84
Surface Sand	3.30 m	1818	6	8.3	5.25	1.4	0.16	0.13	4.45	0.95	0.02	0.06	0.64
Surface Sand	4.0 m	4411	7	8.1	13.5	3.6	0.26	0.5	15.4	0.99	0.25	0.25	1.54
Oxidized Till	1.5 m	1817	8	8.3	70.0	64.9	0.65	4.5	118.0	0.95	0.55	0.36	9.40
Oxidized Till	2.25 m	WTW#1	9	8.0	34.5	39.5	0.60	0.5	74.5	0.5	0.30	0.33	5.80
Oxidized Till	2.25 m	1817	10	8.2	33.0	58.3	0.43	6.5	81.2	0.67	5.41	0.19	7.40
Oxidized Till	2.45 m	4412	11	7.8	34.5	31.6	0.63	0.13	65.8	0.50	0.02	0.22	5.20
Oxidized Till	3.0 m	1816	12	8.0	42.5	50.3	0.69	0.70	93.0	0.60	0.13	0.33	7.20
Oxidized Till	4.5 m	1814	13	8.0	46.5	53.9	0.59	2.0	99.2	0.54	0.02	0.22	7.60
Oxidized Till	5.0 m	1817	14	7.8	35.5	45.7	0.51	1.2	76.0	0.62	7.63	0.22	6.60
Oxidized Till	6.0 m	1814	15	7.9	44.0	50.0	0.60	1.2	93.8	0.53	0.14	0.19	7.20
Oxidized Till	6.75 m	WTW#1	16	8.0	20.5	41.6	0.49	0.70	58.0	0.65	1.51	0.19	5.30
Oxidized Till	10.0 m	4412	17	8.0	46.5	52.0	0.59	0.13	100.5	0.56	0.01	0.17	7.40
Oxidized Till	13.0 m	4413	18	7.8	38.5	28.3	0.59	0.70	58.0	0.58	7.98	0.28	5.20
Buried Sand	7.5 m	1817	19	7.9	42.5	50.6	0.58	2.2	69.2	0.55	22.81	0.22	7.60

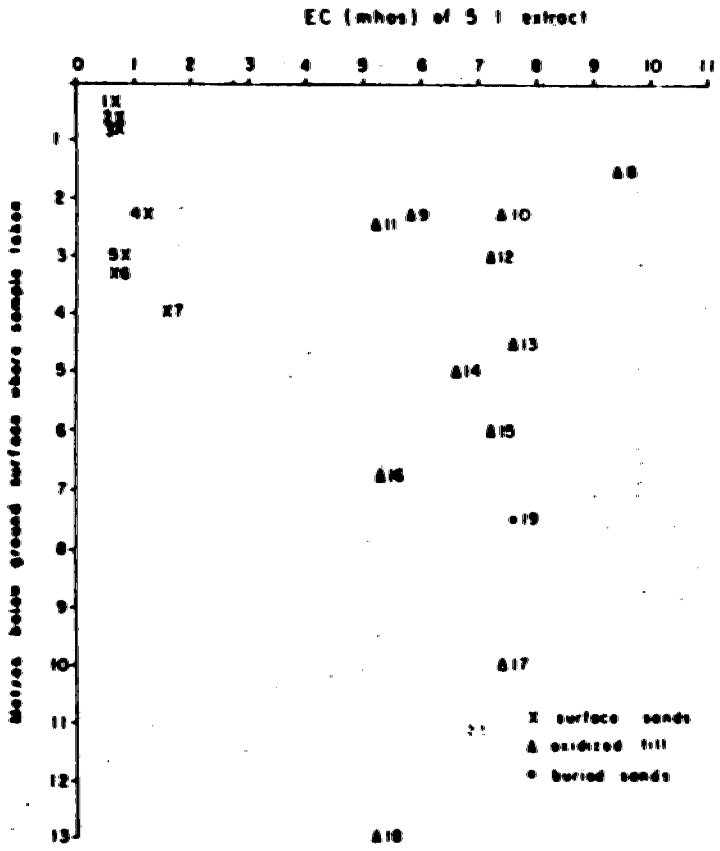


FIGURE 20a: Electrical Conductivity of 5:1 Aqueous Extracts.

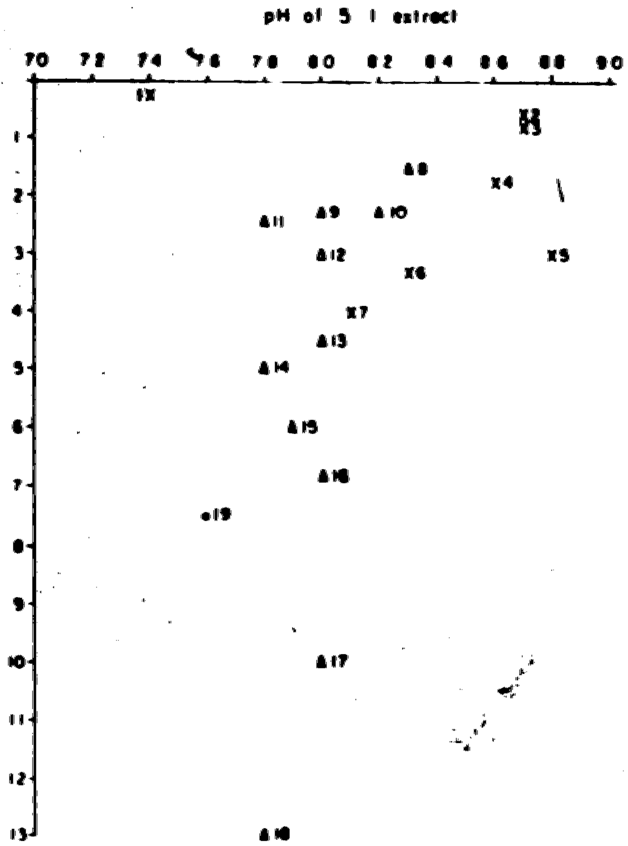


FIGURE 20b: pH of 5:1 Aqueous Extracts.

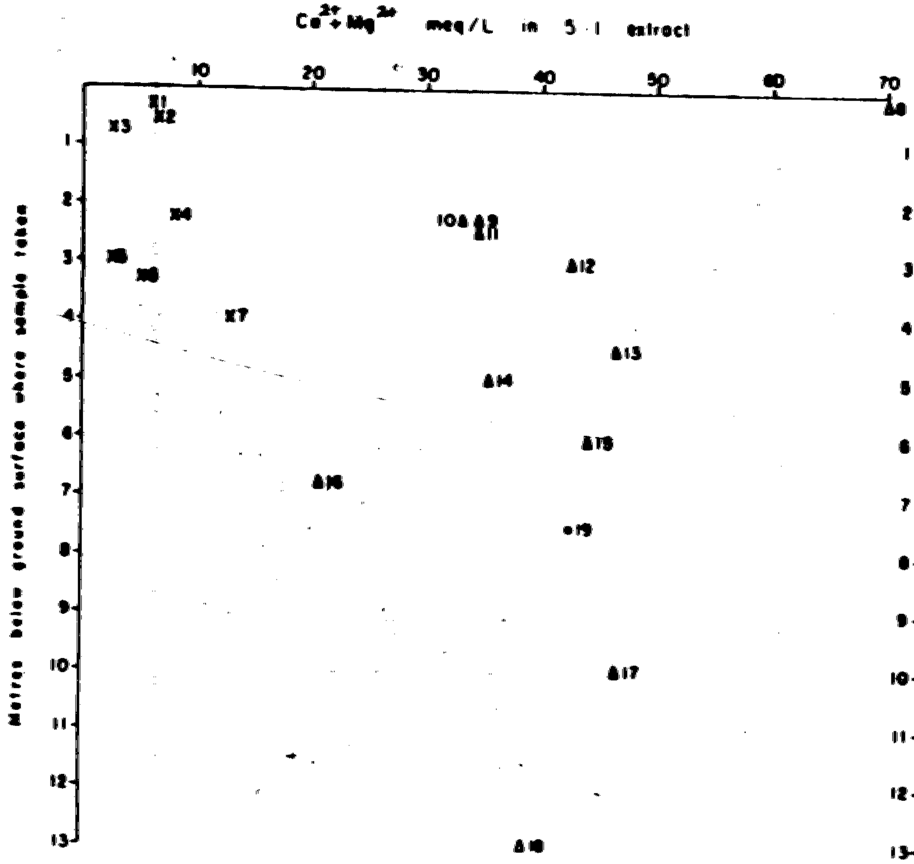


FIGURE 20c: Calcium and Magnesium Concentration in 5:1 Aqueous Extracts.

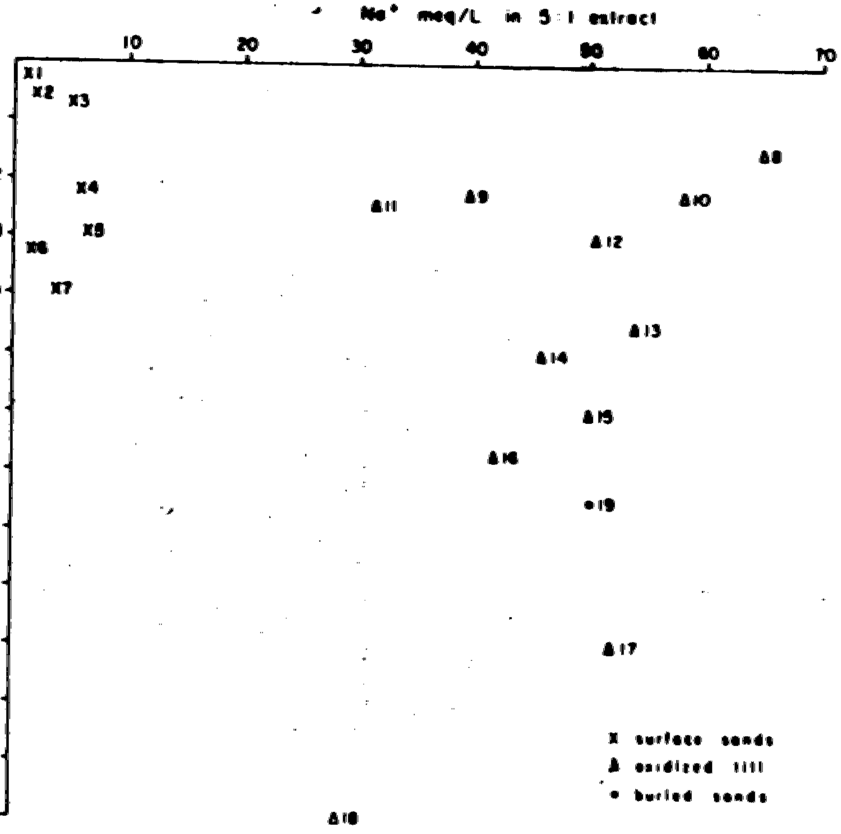


FIGURE 20d: Sodium Concentration in 5:1 Aqueous Extracts.

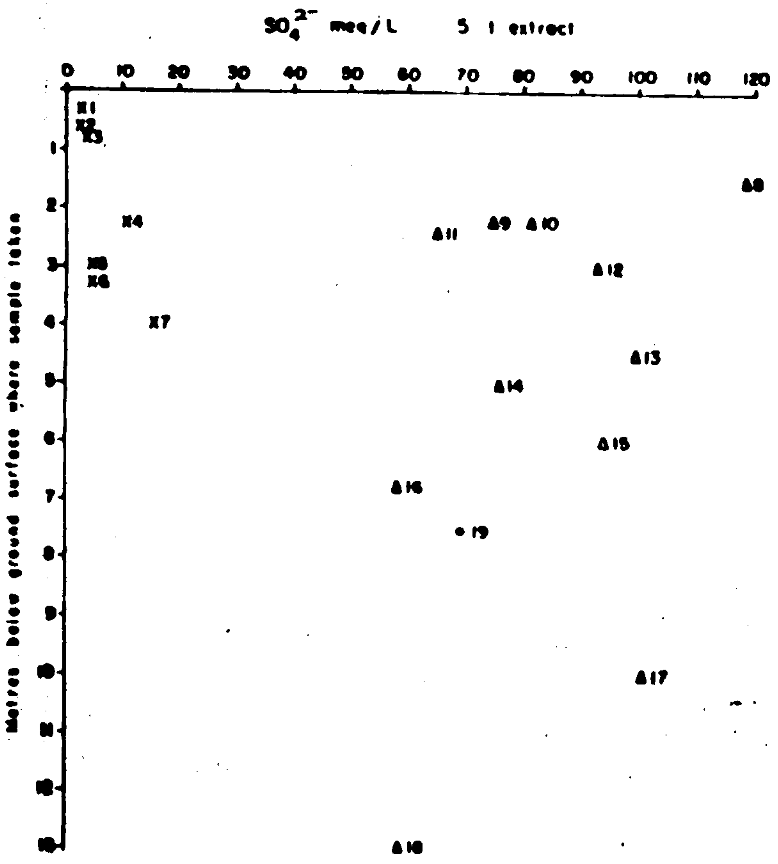


FIGURE 20e: Sulphate Concentration in 5:1 Aqueous Extracts.

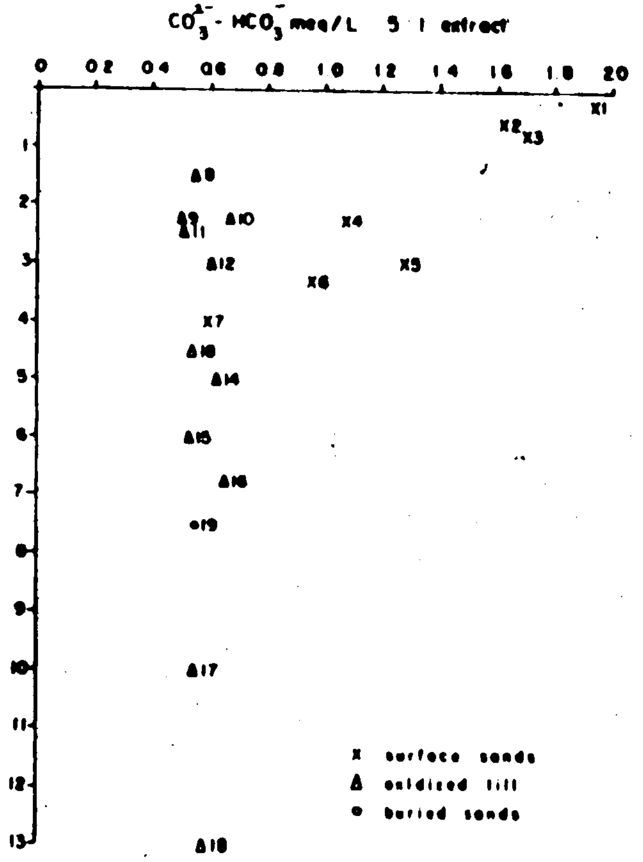


FIGURE 20f: Carbonate-Bicarbonate Concentration in 5:1 Aqueous Extracts.

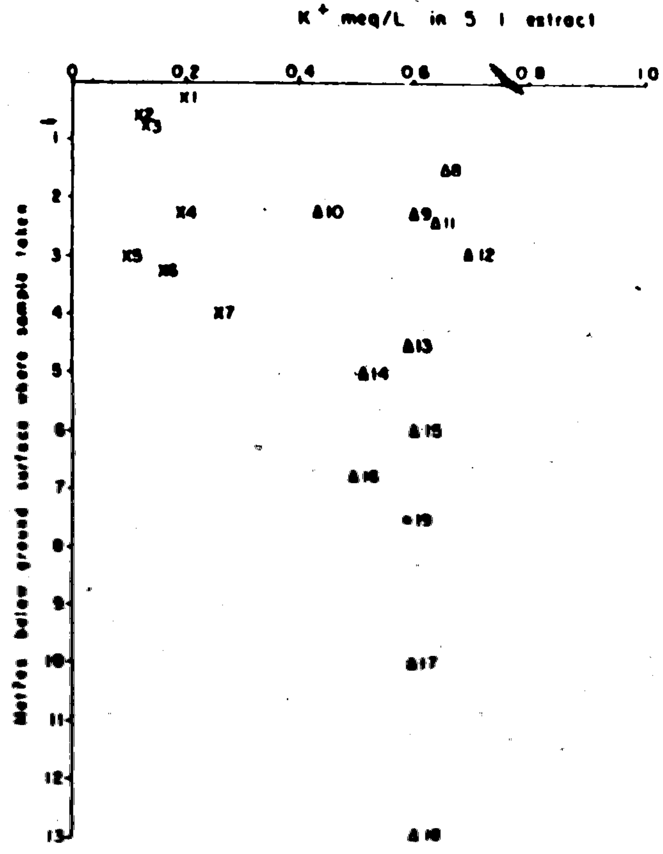


FIGURE 20g: Potassium Concentration in 5:1 Aqueous Extracts.

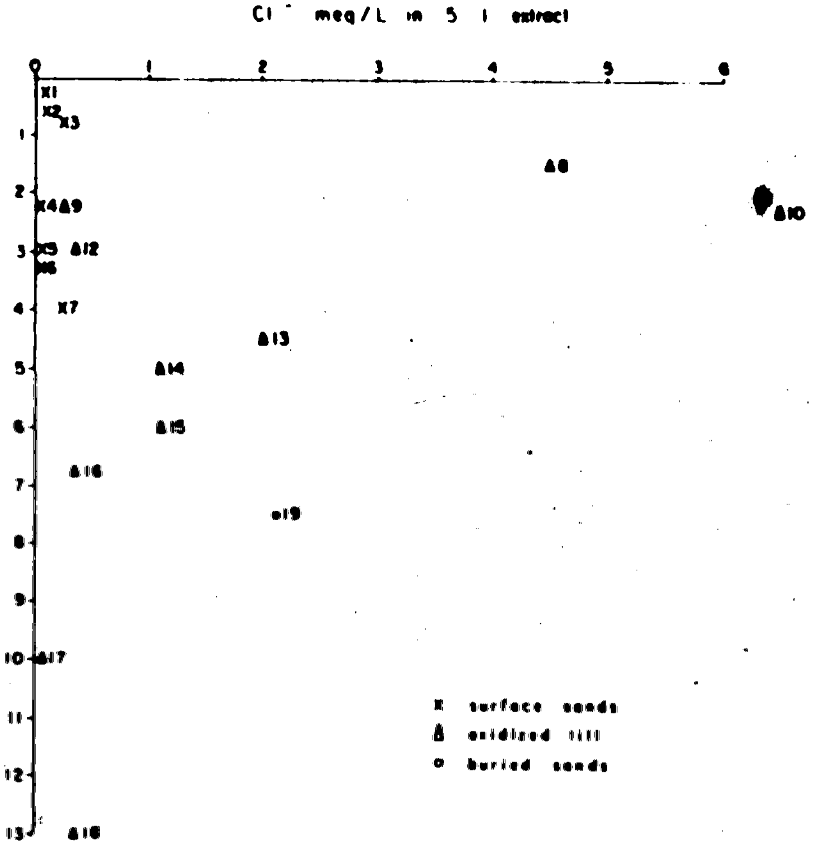


FIGURE 20h: Chloride Concentration in 5:1 Aqueous Extracts.

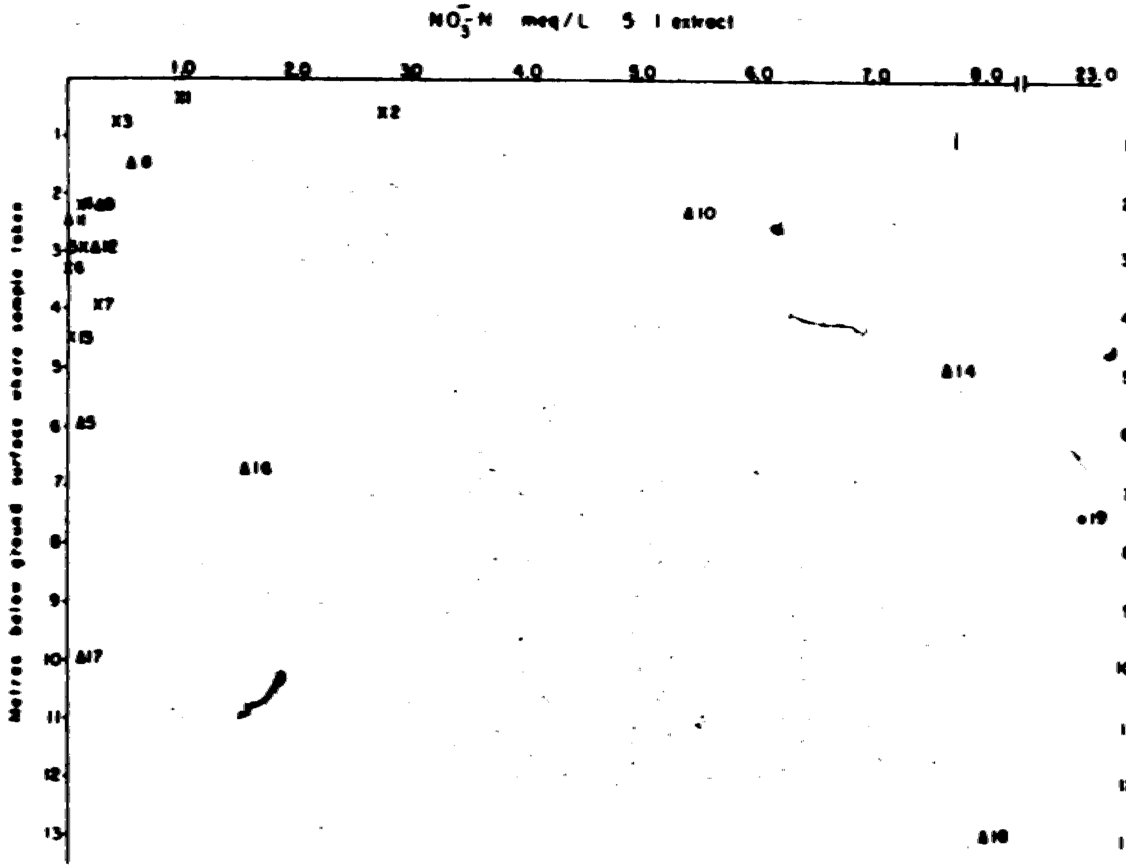


FIGURE 20i: Nitrate-Nitrogen Concentration in 5:1 Aqueous Extracts.

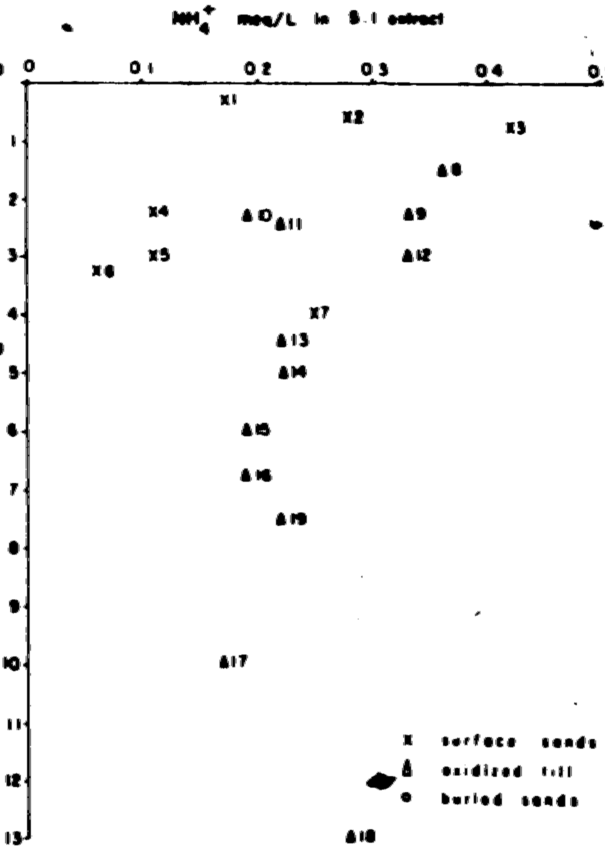


FIGURE 20j: Ammonium Concentration in 5:1 Aqueous Extracts.

samples are considerably greater than those from sands. The mean electrical conductivity of till extracts is 6.75mhos (s.d.=1.30mhos), while the mean electrical conductivity of the sand extracts is 0.93 mhos (s.d.=0.34mhos). Sample 19 however, seems to be an exception. The position and nature of this buried sand sample (19) may account for its exceptional properties. Unlike all the other sand samples which were taken from dry or slightly moist surface sands, sample 19 was taken from a saturated sand lens at a depth of approximately 7.5 m which is overlain by 5 metres of oxidized till. Therefore this sand has probably been subjected to the same processes of salt deposition as the oxidized till around it. Results of water sample analyses (tables 4 to 6), show that water in this buried sand lens has high concentrations of a number of ions. Therefore when the sample was air-dried in preparation for the extract analysis, any salts present in the saturated sample would have been precipitated. Once water was added to the sample to obtain the 5:1 extract these salts would be redissolved and be detected in the analysis of the water extract from the sample.

The following observations can be made from figures 20(a) to 20(j);

1. The pH of 5:1 extracts falls dominantly in the range 7.8 to 8.8 (figure 20(b)). The mean pH of suface sand extracts is 8.37 (s.d.=0.49). This is slightly greater than the mean ph of the oxidized till extracts which is 7.98

(s.d.=0.16).

2. Extracts from the oxidized till have greater concentrations of calcium and magnesium (figure 20(c)), sodium (figure 20(d)), sulphate (figure 20(e)), and potassium ions (figure 20(g)) than extracts from the surface sands. Mean calcium and magnesium concentration in surface sand extracts is 6.28meq/L (s.d.=3.8meq/L) and 40.54meq/L (s.d.=12.3meq/L) for the oxidized till extracts. Sodium concentrations have a mean of 3.58meq/L (s.d.=2.6meq/L), in the surface sands and 46.92meq/L (s.d.=11.02meq/L) in the oxidized till extracts. Mean sulphate concentrations in the surface sands and oxidized tills are 6.16meq/L (s.d.=4.92meq/L), and 83.53meq/L (s.d.=19.32meq/L) respectively while mean potassium ion concentrations are 0.16meq/L (s.d.=0.05meq/L) in the surface sand extracts and 0.58meq/L (s.d.=0.07meq/L) in the till extracts.

3. Chloride ion concentration (figure 20(h)) shows a similar pattern to that described above except that the division between sand and till extracts is not as marked. Extracts from the surface sands have a mean chloride ion concentration of 0.25meq/L (s.d.=0.17meq/L) while in extracts from the oxidized till the mean is 1.66meq/L (s.d.=2.02meq/L).

4. Figure 20(f) shows carbonate-bicarbonate concentrations in the 5:1 aqueous extracts. In this case sand extracts have greater concentrations than those from the tills which cluster around 0.6 meq/L and are independent

of depth.

Mean carbonate-bicarbonate concentration in the surface sand extracts is 1.3meq/L (s.d.=0.47meq/L) while in the oxidized till extracts the mean is o.57meq/L (s.d.=0.06meq/L). There appears to be a decreasing concentration of carbonate-bicarbonate with depth in the sand extracts.

5. Similarities in the grouping of samples can be seen when the pH graph (figure 20(b)) is compared with the carbonate-bicarbonate graph (figure 20(f)). This is not surprising since carbonate-bicarbonate solution buffers the pH of the system. The reasons for the anomalous behaviour of sample 1 are unclear and may be the result of experimental error.

6. Figure 20(i) shows variation in nitrate-nitrogen concentration with depth in the 5:1 extracts. Mean nitrate-nitrogen concentration is 2.15meq/L (s.d.=3.21meq/L) in the oxidized till extracts and 0.66meq/L (s.d.=0.97meq/L) in the surface sand extracts. The large standard deviations reflect the marked variability of nitrate-nitrogen concentration. Although mean nitrate-nitrogen concentration is greater in the oxidized till extracts than in the those from the surface sands, figure 20(i) shows that there is not the distinct difference betwen extracts from the two units that is observed for other ionic species.

7. Ammonium ion concentrations in the 5:1 extracts are given in figure 20(j). Mean ammonium ion concentration in

the oxidized till extracts is 0.25meq/L (s.d.=0.07meq/L), while in the sand extracts it is 0.20meq/L (s.d.=0.13meq/L). There is little difference between ammonium ion concentrations in extracts from the oxidized till and surface sands.

The results of the 5:1 extract analyses will be used in the discussion to aid in the development of a hydrochemical evolution model for the groundwater system beneath the study site.

5. Interpretation and Discussion of Results

The chemistry of a water sample from any point in the groundwater flow system represents the end product of physical and chemical processes and process interactions which have affected the groundwater, (Schwartz and Domenico, 1973). The order in which the processes occur depends upon the direction of groundwater flow and the physical, chemical and biochemical conditions existing within the flow system. The relative magnitude of process and effect also influences the chemical quality of the water sample.

In an attempt to better understand the results of the chemical analyses of water samples from beneath the study site, a number of processes, which have been identified in previous studies of groundwater flow systems through overburden deposits (Moran et al., 1978a,1978b; Hendry, 1981), will be examined here. These processes cannot be totally separated from each other since they interact considerably, therefore the nature and consequences of their interactions will also be considered. Once the major processes and the results of their interactions have been discussed in general terms, they will then be considered with respect to the observed changes in chemical quality as water passes through the overburden deposits beneath the study site. In this way observed changes in groundwater chemical quality will be explained in terms of the processes causing the observed response.

Finally, the hydrochemical evolution model which best explains the observed changes in chemical quality of groundwater as it passes through the overburden deposits will be presented.

5.1 Processes Occurring in the Groundwater Flow System

The major processes which modify the chemical quality of the overburden deposits will be considered here.

5.1.1 Soil zone CO_2 production and dissolution

Respiration of micro-organisms and the oxidation of organic matter by aerobic bacteria produce carbon dioxide in the root zone. The oxidation reaction is described by the equation;

$$O_2(g)+CH_2O \longrightarrow CO_2(g)+H_2O(l) \qquad (10)$$

where CH_2O represent the basic unit of organic matter. This reaction leads to partial pressures of carbon dioxide (pCO_2) in the soil atmosphere from 10^{-3}atm to 10^{-1}atm (Freeze and Cherry, 1979), which are greater than atmospheric partial pressure of carbon dioxide at $10^{-3.5}$atm.

The solubility of carbon dioxide in water is inversely related to temperature. When carbon dioxide dissolves in water, carbonic acid is formed which further dissociates into hydrogen cations and bicarbonate anions according to the reaction;

$$H_2O(l)+CO_2(g) \rightleftharpoons H_2CO_3 \rightleftharpoons H^+ + HCO_3^- \qquad (11)$$

In this way water infiltrating into the soil zone and percolating down through it dissolves carbon dioxide and becomes increasingly acidic because hydrogen ions are produced. Moran et al. (1978b) calculate that water brought into contact with a gas phase where the partial pressure of carbon dioxide is 10^{-2}atm will have an equlibrium pH of approximately 5.0 and if the partial pressure of carbon dioxide is increased to 10^{-1}atm, the equlibrium pH would be 4.2 assuming no other buffering reaction occurred. For this reason Freeze and Cherry (1979) refer to the soil zone as an 'acid pump' in the groundwater flow system. This process can therefore account for the partial pressures of carbon dioxide with which the water samples from the study site are in equilibrium (table 13) which are greater than values which would be produced through equilibrium with atmospheric carbon dioxide partial pressure.

5.1.2 Pyrite (FeS_2) oxidation

Pyrite (iron sulphide) oxidation is another process which increases the acidity of water passing through the shallow zones of the flow system. This process has been identified in areas of overburden deposits similar to those at the study site (Moran et al., 1978a). Pyrite was not directly observed within the tills of the study area although it may have been present in a disseminated form. However, Hendry (pers. comm.) has found disseminated pyrite in oxidized till from the Enchant area of southern Alberta during chemical analysis of oxidized till samples.

The pyrite oxidation process uses atmospheric oxygen which is dissolved in the percolating water; it proceeds according to the equation;

$$4FeS_2(s)+15O_2(g)+14H_2O(l) \longrightarrow 4Fe(OH)_3(s)+16H^{+}+8SO_4^{2-} \qquad (12)$$

(Moran et al., 1978b). Since a large number of hydrogen ions are produced in this reaction, only a small amount of oxidation is required to significantly increase the acidity of the surrounding water. The pH attained by the water and the concentration of sulphate (SO_4^{2-}), ions released by the oxidation of pyrite are dependent upon the avalibility of pyrite, the degree of saturation of the deposits, and whether the system is open or closed with respect to availible oxygen. In an open system, depleted dissolved oxygen concentrations are replenished from the atmosphere and subsequently used for the oxidation reaction. Hence, pyrite oxidation continues, pH declines and sulphate ion concentrations increase. Moran et al. (1978b) demonstrate that under open system conditions, where dissolved oxygen used in the oxidation reaction is continually replenished, and where the porous medium is saturated, water with a minimum pH of 2 and a maximum sulphate ion concentration of 450mg/L can be obtained through pyrite oxidation. However such ideal conditions rarely exist in natural groundwater flow systems through overburden deposits. In the glacially derived deposits examined by Moran et al. (1978a) pyrite oxidati[illegible]urred in the unsaturated zone as water infiltrated toward the water table. In this case rain and snow melt contained only enough dissolved oxygen to produce 17mg/L of sulphate ions. The field situation described here is similar to the situation described by Moran et al. and therefore probably produces sulphate ions in concentrations close to 17mg/L. Some other process must therefore be invoked to account for the high sulphate ion concentrations observed at or just below the water table over much of the study site.

If some other process occurs in the groundwater flow system, simultaneously with pyrite oxidation, and uses the hydrogen ions which are being produced, the pH of the water will not decline to the low levels possible if pyrite oxidation were taking place alone. Dissolution of carbonates is such a process and will be discussed in the next section.

5.1.3 Dissolution and precipitation of carbonates

The two most common carbonates which occur in overburden deposits are calcite ($CaCO_3$), and dolomite ($CaMg(CO_3)_2$). Carbonates were identified in the tills of the field area by acid tests both as clasts and in the disseminated form in the till matrix. These carbonates were assumed to be dominantly calcite and dolomite because of the frequency of water sample saturation with respect to these minerals. However, it is also recognised that contributions of calcium, magnesium and bicarbonate ions from other sources could produce positive saturation indices for calcite and dolomite through the common ion effect.

For calcite and dolomite to dissolve in the groundwater it must be acidic because the dissolution reactions require the presence of hydrogen ions;

$$CaCO_3(s)+H^+ \longrightarrow Ca^{2+}+HCO_3^- \qquad (13)$$

$$CaMg(CO_3)_2(s)+2H^+ \longrightarrow Ca^{2+}+Mg^{2+}+2HCO_3^- \qquad (14)$$

Hence, the dissolution of carbonates is aided by soil production and dissolution of carbon dioxide and by pyrite oxidation, which both produce hydrogen ions. The two reactions illustrated above (equations 13 and 14), buffer the pH of the groundwater by using H^+, and prevent it from becoming exceptionally acidic.

Dissolution of carbon dioxide is the major producer of hydrogen ions in most groundwater systems, (Freeze and Cherry, 1979). The increasing solubility of carbon dioxide with decreasing temperature has already been outlined. This effect is transmitted to the solubility of carbonates through the availibility of hydrogen ions for the dissolution reactions (Freeze and Cherry, 1979). Hence carbonate solubility is also inversely related with temperature.

Since temperature of the groundwater affects its ability to dissolve carbonates, changes in temperature through the year will result in temporal variation in carbonate solubility. In winter or early spring, cold water entering the flow system from the surface is warmed as it moves along the flow path; thus its ability to dissolve carbon dioxide, and consequently carbonates, progressively decreases. Carbonates which were taken into solution at lower temperatures are therefore precipitated. Conversely, in the summer warm water penetrates the soil zone and temperature decreases along the flow path thereby increasing the solubility of carbon dioxide and also carbonates (Freeze and Cherry, 1979).

At carbon dioxide partial pressures typical of those in the soil atmosphere (10^{-3}atm to 10^{-1}atm), Freeze and Cherry (1979) calculate that dissolution of calcite and dolomite to equilibrium will produce a bicarbonate ion concentration in the range of 100mg/L to 600mg/L under open system conditions where CO_2 is continually replenished. Moran et al. (1978a) calculate that pure water at 10 degrees celsius containing carbon dioxide dissolved to equilibrium at partial pressures in the range $10^{-2.5}$atm to 10^{-1}atm will, under open system conditions, produce water with an equlibrium pH of 6.6 to 7.7 and bicarbonate ion concentrations in the range 192mg/L to 610mg/L. In a closed system however, where carbon dioxide is not consistently replenished, bicarbonate ion concentrations will not be this high.

Shallow samples from the field area often had pH's and bicarbonate ion concentrations within or slightly higher than the range predicted by Moran et al.(1978a) and Freeze and Cherry (1979) (tables 4 to 6). They were also commonly saturated or supersaturated with respect to calcite and dolomite, (table 12). The slightly higher bicarbonate concentrations than predicted may result from water temperatures lower than 10 degrees celsius which, as already outlined, would increase carbonate solubility.

Deeper water samples from the field area, especially those from the oxidized till (tables 4 to 6), often have bicarbonate ion concentrations considerably greater than the equilibrium values predicted by Moran et al. (1978a) and Freeze and Cherry (1979). Also, saturation indices with respect to calcite and dolomite for water samples from the oxidized till are mostly positive, indicating supersaturation, (table 12). Further consideration of the table of saturation indices reveals that saturation of the water samples with respect to calcite is reached in the surface sands, hence increased calcite dissolution does not appear to be the mechanism by which bicarbonate ion concentrations increase through the profile (figure 19). Increased dissolution of dolomite cannot be invoked either since dolomite saturation levels vary little between the surface sands and the oxidized till (table 12). Some other process therefore seems to be producing bicarbonate ions within the oxidized till unit.

When calcite and dolomite are both present in a till, the temperature of the groundwater and the order in which the groundwater encounters the minerals, will determine its resulting chemical quality. If the groundwater encounters calcite first and dissolves it to equilibrium then some dolomite can still be dissolved since the water must attain certain Mg^{2+} levels in order that the ion activity product is equal to the equlibrium constant for dolomite at the temperature and pressure of the system. However, when more dolomite is dissolved, Ca^{2+} and HCO_3^- concentrations also increase (equation 14) and this can cause the precipitation of calcite through the common ion effect. Such a chain reaction will cause a decrease in the Ca:Mg molar ratio and may account for some of the low Ca:Mg ratios observed at depth in the field area which could not be attribited to evaporation and evapotranspiration effects.

In the case where dolomite is the first carbonate mineral encountered by the groundwater and is dissolved to equilibrium, the temperature of the water is very important (Freeze and Cherry, 1979). Through comparison of equilibrium constants for calcite and dolomite at various temperatures and at 1 bar total pressure, Freeze and Cherry show that if groundwater at 10 degrees celsius dissolves dolomite to saturation and then encounters calcite, saturation with respect to calcite also occurs through the common ion effect and no further calcite dissolution is possible. However, if the water dissolves dolomite at a temperature lower than 10 degrees celsius, it becomes supersaturated with respect to calcite and precipitation occurs. Conversely, if water dissolves dolomite to saturation at temperatures above 10 degrees celsius and then encounters calcite, more calcite can be dissolved and the water becomes supersaturated with respect to dolomite.

In tills, calcite and dolomite often exist together and simultaneous dissolution occurs (Freeze and Cherry, 1979). The chemistry of the water which results from such a dissolution mechanism is dependent upon the relative abundance of each mineral and their relative rates of dissolution which are, in turn, controlled by temperature and partial pressure of carbon dioxide. For this reason, Ca:Mg molar ratios can develop over a wide range from less than one to more than one (Freeze and Cherry, 1979). Such variations in dissolution reactions can therefore explain the observed variations in Ca:Mg molar ratios (table 14).

The presence of Ca^{2+}, Mg^{2+}, and HCO_3^- ions in solution, which have been derived from sources other than carbonate solution, decreases the solubility of calcite and dolomite through the common ion effect. The total ionic strength of the groundwater also has a significant influence on the solubility of calcite and dolomite. As the total ionic strength of the groundwater increases, the activity coefficients of the major ions in solution decrease (Freeze and Cherry, 1979), (figure 21). This leads to an increase in the solubility of minerals such as dolomite and calcite, since the equlibrium constant is dependent upon the activities of the ions in solution (equation 6).

Partial pressure of carbon dioxide, (pCO_2) variation in water samples from the field area is shown in table 13 and is inconsistent with depth. WATEQF computes the partial

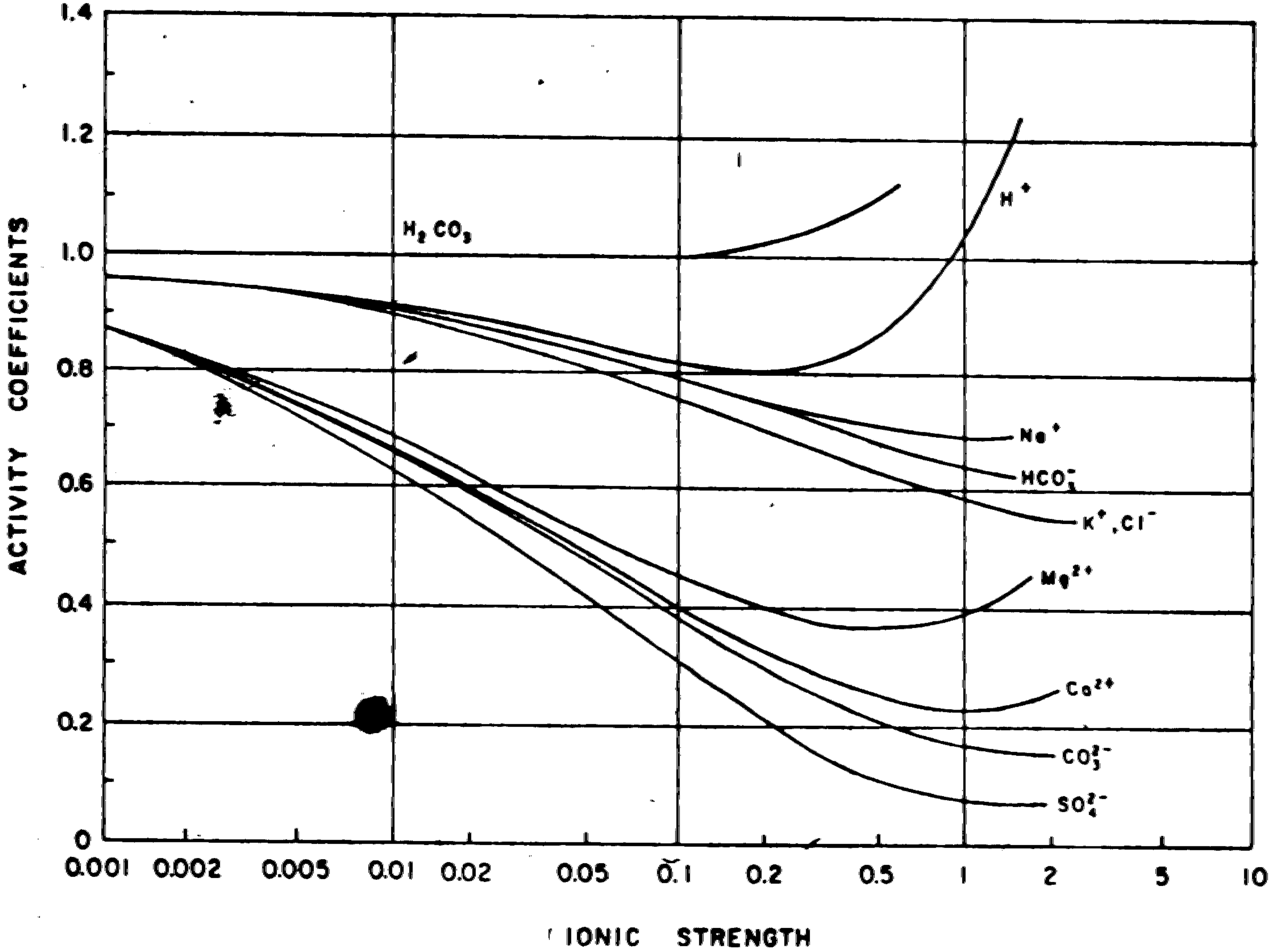


FIGURE 21: Relation of Activity Coefficients of Ions in Solution to Ionic Strength of the Solution (source, Freeze and Cherry, 1979).

pressure of carbon dioxide with which the water samples are in equilibrium, from the from field pH and analytical bicarbonate ion concentration taking into account the temperature of the system. Therefore the high carbon dioxide partial pressure values which are often observed in the oxidized tills are probably a function of the high bicarbonate ion concentrations which have already been noted. It is therefore questionable whether the high partail pressure of carbon dioxide values indicated by WATEQF will result in significantly increased carbonate dissolution since bicarbonate ions are produced by both carbon dioxide and carbonate dissolution and the common ion effect will limit both reactions. If bicarbonate ions are also being produced by some other reaction, the precipitation of calcite and dolomite may occur through the common ion effect even though computed carbon dioxide partial pressure indicates that increased carbonate dissolution should be occurring.

5.1.4 Dissolution and precipitation of sulphates

The dissolution of sulphates, especially gypsum ($CaSO_4.2H_2O$) and anhydrite ($CaSO_4$) has been invoked by Moran et al. (1978a, 1978b) as the most reasonable mechanism to account for the high sulphate concentrations observed in groundwater from overburden deposits. Gypsum dissolves in water quite quickly and laboratory experiments by Kemper et al. (1975) have shown that equilibrium is normally reached within minutes or hours. The solution of gypsum is described by the equation;

$$CaSO_4.2H_2O \rightarrow Ca^{2+}+SO_4^{2-}+2H_2O \qquad (15)$$

Cherry (1968) found that if gypsum is dissolved to equilibrium in deionized water at 8 degrees celsius, 520mg/L of calcium ions and 1240 mg/L of sulphate ions would be produced. However in water containing other ions the solubility would increase in a similar way to that described for carbonates because of decreased activities of ionic species (figure 21).

The origin of gypsum in the overburden deposits may have been by the squeezing of brackish water into the deposits during glacial loading (Cherry, 1972). However Moran et al. (1978a) state that there is a more reasonable way to account for the presence of gypsum in the shallow overburden deposits. Oxidation of pyrite in the shallow zones, where atmospheric oxygen is readily availible, produces sulphate ions, while calcite and dolomite dissolution in the presence of high carbon dioxide partial pressures produce calcium ions. Subsequent excessive evaporation and evapotranspiration from the unsaturated zone which often occurs in a semi-arid area such as the study site, results in the precipitation of gypsum and calcite in the unsaturated zone. This gypsum is then availible for dissolution in water which subsequently infiltrates through this zone during groundwater recharge events. Pyrite oxidation alone cannot account for the very high sulphate concentrations which exist in water samples from the field area. Therefore shallow gypsum precipitation during dry spells and solution during recharge events must be invoked. This idea is supported by the attainment of supersaturation with respect to gypsum at a number of shallow piezometers across the study site (table 12).

Gypsum dissolution can limit the amounts of calcite or dolomite which are able to dissolve, by the common ion effect. Water which encounters and dissolves gypsum at shallow depth will only be able to dissolve limited quantities of calcite or dolomite since calcium ions are already present in the water. Saturation indices with respect to calcite and dolomite (table 12) show that samples are often saturated or supersaturated with respect to these two minerals. Hence, the common ion effect through gypsum dissolution is not preventing saturation with respect to calcite or dolomite.

Other sulphate minerals such as mirabillite ($Na_2SO_4.10H_2O$), epsomite ($MgSO_4.7H_2O$) and glauberite ($Na_2Ca(SO_4)_2$), may be contributing to the high sulphate concentrations in the shallow groundwater. However the effect of these is difficult to assess since no sulphates other than gypsum were shown in the WATEQF output as being present in the groundwater samples at concentrations close to saturation. Therefore gypsum, which was often at saturation in the water samples, seems to be the dominant sulphate in this flow system.

Another alternative for a sulphate source is the input of sulphate to the flow system from atmospheric sources, (pers. comm. F.Schwartz). Sulphate ion concentration in precipitation will only be of the order of a few milligrams per litre, however, over geologic time sulphate deposits could build up to significant concentrations in the unsaturated zone especially with the dominance of evaporation and evapotranspiration in this region of Alberta.

5.1.5 Cation exchange processes

The occurrence of cation exchange in overburden deposits with a large proportion of colloidal particles and the mechanism by which this process occurs, have already been outlined. In this section the effects of cation exchange on the quality of groundwater as it passes through the flow system will be discussed.

The most important ions in the groundwater which take part in cation exchange reactions are calcium (Ca^{2+}), magnesium (Mg^{2+}), and sodium (Na^{+}). Calcium and magnesium ions are preferentially adsorbed onto clays in exchange for sodium ions, according to the equation;

$$Ca^{2+}(aq)+Mg^{2+}(aq)+Na\text{-}clay \rightleftharpoons 4Na^{+}+CaMg\text{-}clay \qquad (16)$$

This reaction is reversible and depends on the relative concentrations of adsorbed and aqueous species, however the forward reaction usually occurs due to the stronger adsorption affinities of calcium and magnesium than that of sodium. When the forward reaction occurs, the groundwater becomes enriched with sodium ions and depleted in calcium and magnesium ions, assuming all are present. Since each mole of calcium or magnesium ions which is removed from the water is replaced by 2 moles of sodium ions, cation exchange causes a slight increase in the total dissolved solids concentration of the groundwater.

The major source of calcium and magnesium ions which enter into the cation exchange reactions appears to be the dissolution of calcite, dolomite and gypsum. As calcium and magnesium ions are removed from the groundwater by cation exchange, more of these ions must be produced to maintain the ion activity product of each species. A secondary effect of cation exchange is, therefore, to increase the dissolution of calcite, dolomite and gypsum through the common ion effect. Kinetic factors such as temperature and pressure of the system control the rates at which these three minerals dissolve in response to the cation exchange reaction. At a given temperature and pressure calcite may dissolve more rapidly than dolomite causing a relative enrichment of calcium ions over magnesium ions in the groundwater. In this way variable dissolution of calcite, dolomite and gypsum in response to cation exchange controls the magnitude of ion concentration variation and molar ratios in the groundwater. This is therefore another mechanism which could produce the observed variations in Ca:Mg molar ratios with depth (table 14).

The common ion effect and subsequent response of the system by increased dissolution of calcite, dolomite and gypsum result in two secodary effects:

1. Increased bicarbonate ion (HCO_3^-) concentration since this is produced during the dissolution of calcite and dolomite (equations 13 and 14),

2. Increased pH because hydrogen ions (H^+) are used up in the dissolution process.

The increasing sodium concentrations observed along parts of the groundwater flow path beneath the field area may, therefore, be explained in terms of cation exchange processes. This would be supported by simultaneous decreased calcium and magnesium ion concentrations, assuming these effects are not masked by the common ion effect or other processes occurring in the groundwater flow system.

Moran et al. (1978a) explain high sodium concentrations in water samples from overburden deposits in North Dakota in terms of cation exchange processes, since no soluble sodium salt deposits could be identified which could account for the high sodium concentrations present in the groundwater. The presence of sodium at the exchange sites was attributed to the total immersion of the sediment in an aqueous solution with a large sodium excess. The origins of the sodium rich clays in the study area considered here are not known.

Even though calcium and magnesium are preferentially [illegible]sorbed over sodium at cation exchange sites, the cation exchange reaction described in equation 16 is reversible. In order for the reverse reaction depicted in equation 16 to occur, sodium concentrations in the groundwater must become high enough to overcome the greater adsorption affinities of calcium and magnesium. No reference could be found which examines the relative concentrations of sodium, calcium and magnesium ions in the aqueous phase with respect to the direction of cation exchange reactions. Sodium concentrations at the study site reach almost 2900mg/L (126mmoles/L)(tables 4 to 6) in water samples from the oxidized till unit, when calcium ion concentration in the same sample is approximately 450mg/L (1.1mmoles/L). The order of magnitude difference in aqueous molar concentrations observed here is felt to be sufficient to overcome the difference in adsorption affinities, and therefore to promote the back reaction from right to left in equation 16. The evidence for reverse cation exchange occurring beneath the study area is examined later in this chapter in reference to the results of specific water sample analyses. Hem (1970) states that in irrigated areas the exchange of calcium for sodium in soil moisture may proceed forward or in reverse at different times, and at any particular site may fluctuate extensively. Hence cation exchange processes not only occur in the saturated flow system, but can also affect the quality of the water before it reaches the water table.

5.1.6 Denitrification and sulphate reduction

A number of researchers including Gillham et al.(1978), Hendry (1978), and Trudell (1980) have observed sharp decreases in nitrate-nitrogen concentrations as groundwater passes from shallow to deeper parts of the groundwater flow system. The decreases in nitrate-nitrogen concentrations were found to correspond with decreased dissolved oxygen concentrations and decreasing redox potential (Eh). They were attributed to denitrification by nitrate reducing bacteria.

Certain bacteria such as *Pseudomonas* sp. are able to assimilate nitrate during their respiration process, when the amount of oxygen availible is inadequate to meet their respiratory requirements. The nitrate assimilation reaction by such bacteria was described by Stumm and Morgan (1970) in the equation;

$$4NO_3^- + 5CH_2O + 4H^+ \longrightarrow 2N_2 + 5CO_2 + 7H_2O \qquad (17)$$

The carbon dioxide produced by this reaction then dissolves in the groundwater to form carbonic acid according to the reaction;

$$5CO_2 + 5H_2O \rightleftharpoons 5H^+ + 5HCO_3^- \qquad (18)$$

If equations (17) and (18) are combined the resulting equation for the overall denitrification process is;

$$4NO_3^- + 5CH_2O \longrightarrow 2N_2 + H^+ + 5HCO_3^- + 2H_2O \qquad (19)$$

This shows that more hydrogen ions are produced as a consequence of denitrification than are used, therefore the pH of the groundwater decreases. The bicarbonate concentration of the groundwater increases since these ions are also produced.

The large number of bicarbonate ions produced by the denitrification process will affect any carbonate equilibria existing in the system through the common ion effect. Precipitation of carbonates will result since the increased bicarbonate ion concentration increases the rate of the reactions;

$$Ca^{2+} + HCO_3^- \rightleftharpoons CaCO_3 + H^+ \qquad (20)$$

$$Ca^{2+} + Mg^{2+} + 2HCO_3^- \rightleftharpoons CaMg(CO_3)_2 + 2H^+ \qquad (21)$$

Therefore the observed effects on the groundwater system, which accompany the decrease in nitrate-nitrogen concentrations will be decreased calcium and magnesium ion concentrations, decreased pH, and some increase in bicarbonate ion concentration assuming it is not all used up in the above reactions (equations 20 and 21). In this way calcium and magnesium ion concentrations decrease even though the partial pressure of carbon dioxide computed by WATEQF increases.

The removal of calcium and magnesium ions from the groundwater can also have the effect of limiting cation exchange reactions. However, the secondary effect of increased bicarbonate ion concentration which results from cation exchange processes does not affect the denitrification process because the reaction (equation 19) is not reversible.

Gillham and Cherry (1978) propose that in order for denitrification by nitrate reducing bacteria to occur, dissolved oxygen concentrations must be less than 2mg/L while Broadbent and Clark (1975) indicate that redox potentials must be less than 320mv. However, they also recognise that microsite conditions around the bacteria may be considerably different from those indicated by measuring dissolved oxygen concentrations and Eh at a comparitive macroscale. Hence denitrification may be occurring, even where dissolved oxygen concentrations or redox potentials appear too high.

Dissolved oxygen concentrations in waters from beneath the study site have been seen to range from 0.1mg/L to 4.8mg/L and show little, if any, correlation with depth (figure 16). Many of the samples have dissolved oxygen concentrations less than 2mg/L which, according to Gillham and Cherry (1978), would allow denitrification to occur.

Sulphate reduction is also a bacterial reaction and can be described by the equation;

$$SO_4^{2-}+CH_2O+6H^+ \longrightarrow H_2S+CO_2+3H_2O \qquad (22)$$

(Stumm and Morgan, 1970). However redox conditions must be sufficiently negative (less than -200mv, Bohn et al., 1979), to allow this reaction to proceed. Sulphate reduction would therefore be expected to occur at depth.

Hydrogen sulphide, (H_2S), which is produced during sulphate reduction has a distinct odour of "rotten eggs" which can be detected by humans at concentrations as low as

10^{-8}atm. This is therefore a very useful field indicator of the sulphate reduction process. Conversely, the absence of the hydrogen sulphide odour does not necessarily mean that sulphate reduction is not occuring. The hydrogen sulphide produced may react with iron minerals which are present and so be removed from the system (Bohn et al., 1979).

Apart from decreasing the concentration of sulphate ions in the groundwater, sulphate reduction also results in an increase in the partial presure of carbon dioxide. The carbon dioxide produced in this reaction is dissolved in the groundwater according to the equation;

$$H_2O + CO_2 \rightleftharpoons H^+ + HCO_3^- \qquad (23)$$

By combining equations (22) and (23) the final equation for the sulphate reduction reaction is;

$$SO_4^{2-} + CH_2O + 5H^+ \rightarrow H_2S + HCO_3^- + 2H_2O \qquad (24)$$

The increase in bicarbonate ion concentration which results, has the same effect on carbonate solubility as that described for the denitrification process. In the case of sulphate reduction though, for every five hydrogen ions used in the reaction (equation 24), one bicarbonate ion is produced. Also for each of these bicarbonate ions which is used in the precipitation of carbonates, one hydrogen ion is produced. The net effect is therefore to decrease the number of hydrogen ions in the system and therefore to increase pH.

In summary the major chemical and biochemical processes which are likely to influence the chemistry of the groundwater as it passes through the flow system are

1. CO_2 production in the soil zone; decreases pH. Aids the solution of carbonates.

2. Pyrite oxidation; decreases pH. Aids the solution of carbonates.

3. Dissolution of carbonates; increases Ca^{2+}, Mg^{2+}, and HCO_3^- concentrations. Buffers pH. Causes variations in Ca:Mg molar ratio. Causes precipitation of gypsum through the common ion effect.

4. Precipitation of carbonates; decreases Ca^{2+}, Mg^{2+},and HCO_3^- concentrations. Increases pH. Causes solution of gypsum through the common ion effect.

5. Dissolution of sulphates (gypsum); increases Ca^{2+}, and SO_4^{2-} concentrations. Causes precipitation of carbonates by the common ion effect.

6. Precipitation of sulphates; reduces concentrations of Ca^{2+}, and SO_4^{2-}. May act as a source for the high sulphate concentrations observed.

7. Cation exchange process; increases Na^+ and decreases Ca^{2+} and Mg^{2+} concentrations. Causes increased dissolution of calcite and dolomite. Increases HCO_3^- concentration and pH through the common ion effect. Occasionally, if conditions allow, the reverse reaction occurs with the opposite effects.

8. Denitrification; decreases NO_3^-concentrations. Decreases pH and increases HCO_3^- concentrations. Causes precipitation of calcite and dolomite. Decreases Ca^{2+} and

Mg^{2+} concentrations through the common ion effect.

9. Sulphate reduction; decreases SO_4^{2-} concentrations. Increases HCO_3^- concentration and increases pH. Causes precipitation of calcite and dolomite. Decreases Ca^{2+} and Mg^{2+} concentrations through the common ion effect.

The order in which these processes occur and the relative magnitude of their effects determines the observed chemical quality of the groundwater.

5.2 Interpretation of Water Sample Analysis Results

In an earlier section of this thesis, results of chemical analyses were presented for all three sets of water samples collected during the summer of 1980. General relationships between some of the measured variables were also observed. The results of both the field and laboratory chemical analyses were statistically summarized as means, ranges and sample standard deviations of variables within each of the four overburden units described (tables 8 to 11).

In this section the changes in some of the measured variables and the relationships between them will be examined. They will be discussed in terms of the processes already described. The general trends in mean concentrations of major ion species between the four geologic units, which were illustrated in figure 19 will also be discussed in terms of these processes.

In order to examine the changes in groundwater chemical quality which occur within each of the four geologic units, the results from the first data set will be used. They will be graphically plotted against depth from which the groundwater sample was taken, for each of the major piezometer nests across the study site. A generalized geologic cross section through the overburden deposits beneath each of these nests will also be illustrated on these plots.

The results from the first sample set were chosen since they include field measurements of pH, dissolved oxygen and redox potential (Eh). The results from this sample set were proven not to be significantly different from either of the other two sample sets by testing the means and sample standard deviations of each of the major ion species within each of the four geologic units, against each other using the differences of means t-test.

The dominant groundwater flow direction beneath the study site has already been shown to be vertically downward, (figures 9 to 11). Changes in water chemistry between point source piezometer intake zones at increasing depths beneath the ground surface are therefore taken in most cases to represent changes occurring along the flow path.

5.2.1 Differences between field and laboratory pH

Field pH of the water samples was seen in the results chapter to be consistently greater than laboratory pH (figure 14) however, no correlation was found between the two measurements. It was suggested that the differences between the two pH measurements may be the result of degassing of carbon dioxide from the sample during collection and laboratory analysis.

In the previous section, one of the major processes influencing groundwater quality was seen to be the dissolution of carbon dioxide in the soil zone, at partial pressures up to two orders of magnitude greater than atmospheric partial pressure. Carbon dioxide and therefore bicarbonate ions are also produced by denitrification and sulphate reduction deeper in the saturated zone of the overburden deposits. Carbon dioxide partial pressure values with which the water samples from the field area were in equilibrium were computed by **WATEQF** from field pH and analytical bicarbonate ion concentrations (table 13). These are greater than atmospheric carbon dioxide partial pressure for all water samples from the field area probably because of the processes outlined above. Contact between the groundwater and atmosphere during collection and analysis of these water samples would therefore result in the loss of carbon dioxide from solution as the sample came to equilibrium with the atmospheric partial pressure of carbon dioxide. This degassing would cause a decrease in hydrogen and bicarbonate ion concentrations and subsequntly pH would increase. Therefore degassing of carbon dioxide seems to be a likely cause of higher laboratory pH than field pH.

Degassing of water samples could also result in the secondary precipitation of carbonates if their saturation levels became too high. However, no such precipitation was observed during the analysis of the water samples and so the degassing was not thought to be a significant problem with respect to altering the total chemistry of the sample.

5.2.2 Redox Conditions and Biochemical Processes

Redox potentials of water samples from beneath the field area were measured during the collection of the first set of samples (table 4). A plot of Eh(mv) versus depth from which the groundwater sample was taken (figure 16) shows little correlation between the two variables due to problems in the sampling procedure which have already been outlined. All measured redox potentials are low enough to allow denitrification (less than 320mv), but none reach the low levels necessary for sulphate reduction (less than -200mv).

Dissolved oxygen concentrations in groundwater samples were found to range from 0.1mg/L to 4.8mg/L, showing no correlation with depth (figure 16). A number of water samples had less than 2mg/L dissolved oxygen which would allow denitrification to proceed assuming nitrate reducing bacteria and a suitable organic source were present.

Figure 22 is a plot of dissolved oxygen concentration (mg/L) against nitrate-nitrogen concentration (mg/L) for the first set of samples. All samples below the nitrate-nitrogen detection limit of 0.1mg/L are plotted below the dotted line. This figure illustrates that even though dissolved oxygen conditions would allow denitrification to occur, many of the groundwater samples still have quite high nitrate-nitrogen concentrations. This may be caused by the absence of the bacteria necessary for denitrification. Many of the groundwater samples have nitrate-nitrogen concentrations which are below the laboratory detection limit (0.1mg/L) and some of these samples come from deep piezometers at nests where shallow piezometers have high nitrate-nitrogen concentrations. If the water at depth formerly had a similar chemical composition to the water presently higher in the profile, then the denitrification process may be inferred to account for the loss of nitrate-nitrogen through the profile. Denitrification will be discussed in greater detail when changes through specific profiles are considered.

The biochemical process of sulphate reduction is also related to redox conditions. All of the groundwater samples marked with a circle on figure 16 had the distinct odour of hydrogen sulphide when the piezometer was sampled, indicating that sulphate reduction is occurring beneath the field area. The measured redox potential of these samples were all negative.

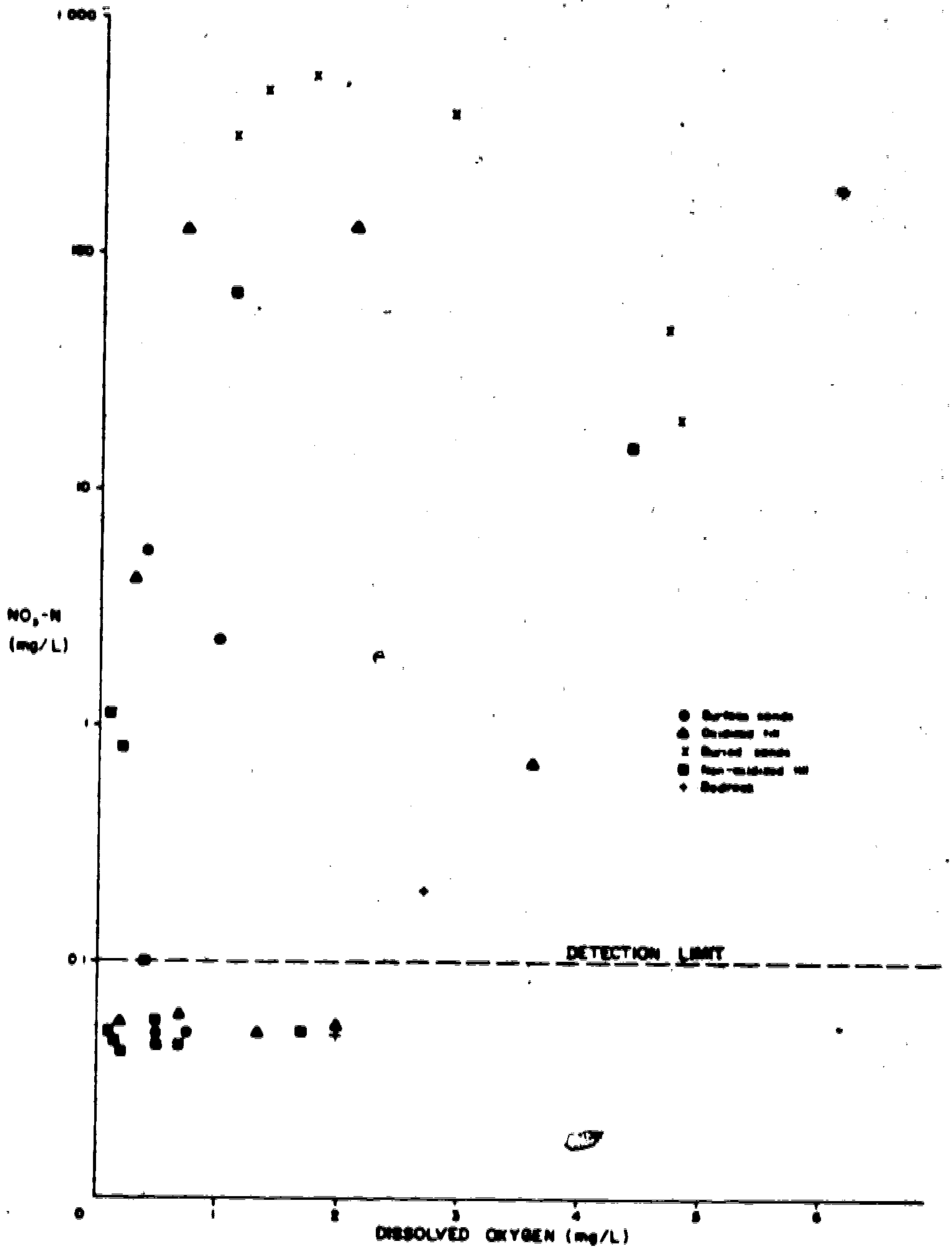


FIGURE 22: Nitrate-Nitrogen Concentration versus Dissolved Oxygen Concentration in Water Samples (1st Sample Set).

When a water sample is taken from its natural environment and brought into contact with atmospheric oxygen, oxidation of reduced species within the water is likely to occur. Hence the Eh measurements here, which were taken with the sample in contact with the atmosphere, are probably considerably higher than the redox potentials which exist *in situ* within the overburden deposits. This explains the apparent reduction of sulphates at redox potentials higher than is normally possible. Better methods of field measurement of redox potential and dissolved oxygen concentration are necessary before total confidence can be placed in results from field measurements.

5.3 Changes in Water Chemistry Between Geologic Units

The changes in mean ionic concentrations of major species in the groundwater, summarized from all three data sets, are illustrated in figure 19. Mean concentration of each species in groundwater from each of the four units is plotted against the mean depth from which groundwater samples in that unit were taken. Concentrations of most major ion species increase from the sands to the oxidized till where they reach their maximum concentration in the profile and then decrease with increasing depth into the buried sands and the non-oxidized till.

Although mean concentrations of all major ion species in the groundwater increase from the surface sands into the oxidized till, the major increases are in sodium (42mmoles/L) and sulphate concentrations (25mmoles/L). In comparison, increases in calcium, magnesium and bicarbonate ion concentrations are quite small at 5.7mmoles/L, 4.0mmoles/L and 6.5mmoles/L respectively. pH of the groundwater decreases between the surface sands and the oxidized till indicating that the concentration of hydrogen ions has increased through some process or combination of processes.

The results of the 5:1 extract analyses showed that the oxidized till contains greater concentrations of water soluble salts than the surface sands, (figures 20(a) to 20(j), table 15), and that calcium and magnesium, sodium and sulphate ions were present in the highest concentrations in the 5:1 aqueous extracts from the oxidized till samples. The increase in ionic concentration of the groundwater from the sands to the tills is therefore not surprising. Even though flow velocities in the oxidized till are so rapid it would appear from considering the mean ionic concentrations that the groundwater has sufficient residence times to dissolve soluble species from the oxidized till. This will be considered further when data from each piezometer nest are examined individually.

Dissolution of gypsum between the sands and the oxidized tills may account for some of the increase in calcium ion concentration in the groundwater however the increase in sulphate ion concentration is greater than that for calcium. The lack of calcium to balance sulphate may be the result of carbonate precipitation induced through the common ion effect by increasing calcium ion concentration. Alternatively, the excess sulphate in solution may be derived from some other source such as dissolution of mirabillite, epsomite or glauberite. The presence of sulphates in the system other than gypsum is difficult to assess since none of the water samples are saturated with any sulphate other than gypsum.

The increase in mean sodium ion concentration in the groundwater is the major contributor to the increase in total dissolved solids between the surface sands and the oxidized till. The till provides an environment where smectite is abundant, potentially allowing cation exchange to occur. This can result in the relative enrichment of sodium ions in the groundwater at the expense of calcium and magnesium, and could account for the large increases in sodium concentration between the surface sands and the oxidized till. It could also explain the lack of balance between calcium and sulphate ion concentration increases.

Calcite and dolomite dissolution increases in response to the loss of calcium and magnesium from the groundwater by cation exchange. This can also explain the slight increase in bicarbonate ion concentration as the groundwater moves between the surface sands and the oxidized tills, since bicarbonate ions are produced during the dissolution process. Calcium and magnesium ion concentrations should only increase enough to replace the ions lost to cation exchange, so that there should not be any change in the observed concentrations of these ions. The observed increase in calcium and magnesium ion concentrations from the surface sand into the oxidized till indicates that calcite and dolomite dissolution is occurring in excess of the amount required to balance the losses from the groundwater to cation exchange sites.

Hydrogen ions are used up during carbonate dissolution (equations 13 and 14); this usually results in increased groundwater pH. In the generalized cross section (figure 19) mean pH of the groundwater decreases between the surface sands and the oxidized till. The process or balance of processes which caused this decrease cannot be inferred from the limited information availible here.

Once the groundwater passes from the oxidized till into the buried sand deposits, mean concentrations of all major ion species except calcium and nitrate-nitrogen decrease; pH increases slightly. It should be noted that mean ionic concentrations for the buried sand deposits were computed from water samples taken at sites 1813, 1814, and 1817. The means plotted on figure 19 are generally lower than the specific ion concentrations at site 1817 because of the influence of the lower concentrations at site 1813 and 1814. Conversely, mean concentrations are greater than the actual concentrations at sites 1813 and 1814. The lower mean ionic concentrations in the groundwater within the buried sands than in the overlying oxidized till are probably a result of chemical or biochemical attenuation processes.

The increase in calcium ion concentrations between the oxidized till and the buried sands, from 391.5mg/L to 433mg/L where all other ionic concentrations except nitrate-nitrogen decrease, is anomalous. When a homogeneity of variance (F-test) and difference of means (t-test) were carried out to compare the mean calcium concentrations, the increase in concentration was found to be insignificant. With the information availible here, no process can be invoked which can explain this increase; it is probably an undesirable consequence of generalizing the data by computing means in order to observe general trends.

The high nitrate-nitrogen concentration in the groundwater samples from the buried sands is a result of including the data from site 1817 when computing the mean. Whether this nitrate-nitrogen is derived from a local geologic source or is the result of agricultural practices is uncertain since information on fertilizer application was not availible from the farmer. However, recent work by Hendry (pers. comm.) has shown that nitrates are present in quite high concentrations in tills from the area, indicating that a geologic source is likely for at least some of the nitrate nitrogen in the groundwater.

Mean concentrations of all ionic species in the groundwater, except bicarbonate, decrease and pH increases slightly between the buried sand and underlying non-oxidized till. This general decrease in mean concentrations is the result of chemical or biochemical attenuation processes such as cation exchange, precipitation, and microbial activity. Precipitation of salts seems likely since crystals were found in samples of the non-oxidized till matrix taken during drilling. These crystals did not respond to acid and were therefore not carbonates, however, they were colourless and soluble in water and may have been sulphates.

Surface areas and times of contact between the groundwater and the till matrix are greater in the unfractured non-oxidized till than in the oxidized till unit where the major paths of water movement are the fractures and the porosity is lower than in the non-oxidized till. The cation exchange process was therefore expected to be more efficient in the non-oxidized till than in the oxidized till. No evidence such as increasing sodium concentration with depth, is observed in figure 19 to support this supposition. However, sodium concentration in the groundwater is often quite high which may induce reverse cation exchange. Such a process would initially cause an increase in calcium and magnesium ion concentrations in the groundwater, although once concentrations became particularly high, precipitation of carbonates and/or gypsum would occur. No large carbonate crystals were found in the non-oxidized till matrix to support this thoery, although testing with acid caused considerable bubbling indicating the presence of disseminated carbonates.

The higher mean bicarbonate ion concentration in the groundwater from the non-oxidized till than that from the buried sands is probably the result of sulphate reduction which is known to occur within this unit. The reaction by which the bicarbonate ions are produced was illustrated in equation (24) earlier in this chapter.

A secondary common ion effect which results from increased bicarbonate ion concentration is the precipitation of carbonates (equations 20 and 21), which in turn causes a decrease in calcium and magnesium ion concentrations in the groundwater. This is probably occurring here, although not to a great enough degree to use up all the bicarbonate ions produced during sulphate reduction. This precipitation of carbonates, in response to sulphate reduction therefore accounts for at least part of the decrease in calcium and magnesium ion concentrations between the buried sands and the non-oxidized tills.

There is a rapid decline in nitrate-nitrogen concentratiion through this part of the profile which is probably the result of denitrification. Even though a number of shallow samples from across the field area have high nitrate-nitrogen concentrations, no sample taken at more than 30m beneath ground surface had a nitrate-nitrogen concentration higher than the standard 10mg/L which is the safe limit for human consumption (Hem, 1970). The denitrification process therefore provides an efficient natural decontamination mechanism. At present, pollution of the groundwater by nitrate-nitrogen, whether of natural or agricultural origin, is not a problem with respect to the vertical movement of water. The effects of lateral water movement through the buried sand deposits will be examined in greater detail later in this chapter.

5.4 Vertical changes in Groundwater Quality Within the Geologic Units

Results of water sample analyses from the first sample set are plotted here against depth and generalized overburden geology for each of the major piezometer nests across the study site, (figures 23 to 29) These figures will be used to illustrate changes in water quality which occur as the water moves within each overburden unit.

The points plotted on these figures represent real data so they have been joined by straight lines. However, this may give a misleading impression of changes in water quality through the deposits. For example, considering the cation plot for site 1813 (figure 25(a)) sodium concentration in the groundwater appears to increase through the buried sands and into the non-oxidized till. If the rate of increase in groundwater sodium concentration observed between the two piezometers in the buried sands is extrapolated to their base, sodium concentration would actually decrease between the upper boundary of the non-oxidized till and the sample point within it.

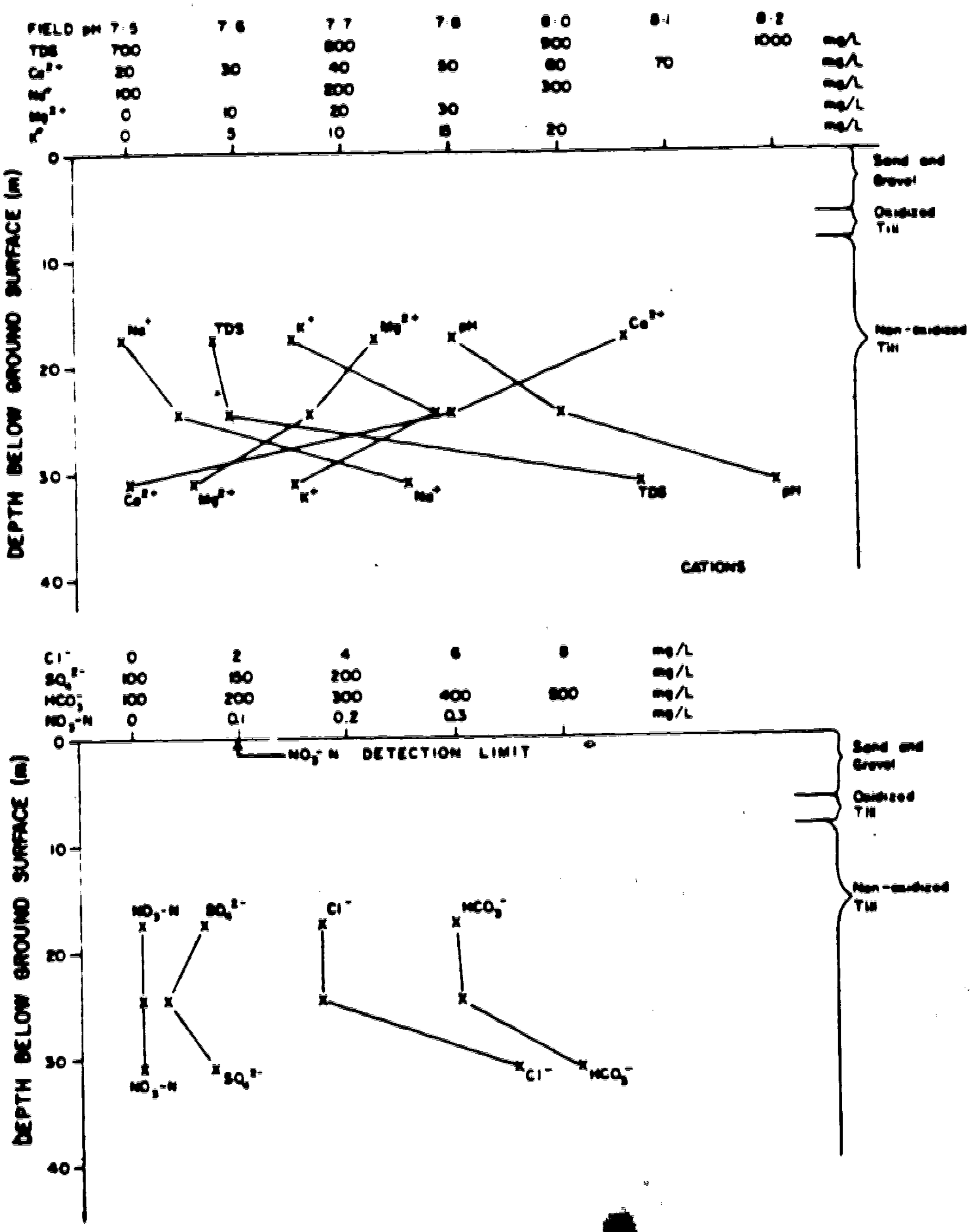


FIGURE 23: Changes in Groundwater Chemistry Beneath Piezometer Nest 1811.

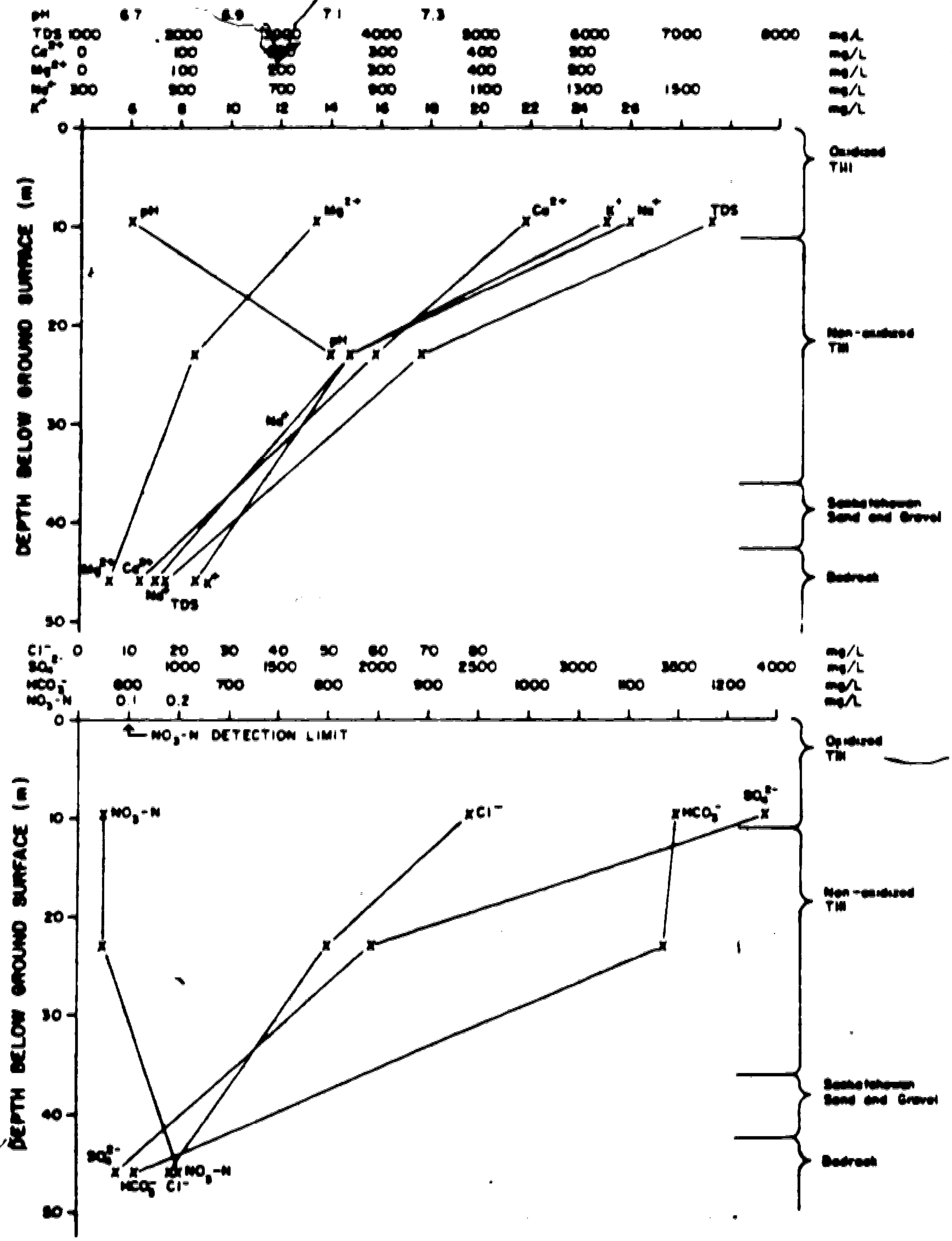


FIGURE 24: Changes in Groundwater Chemistry Beneath Piezometer Nest 1812.

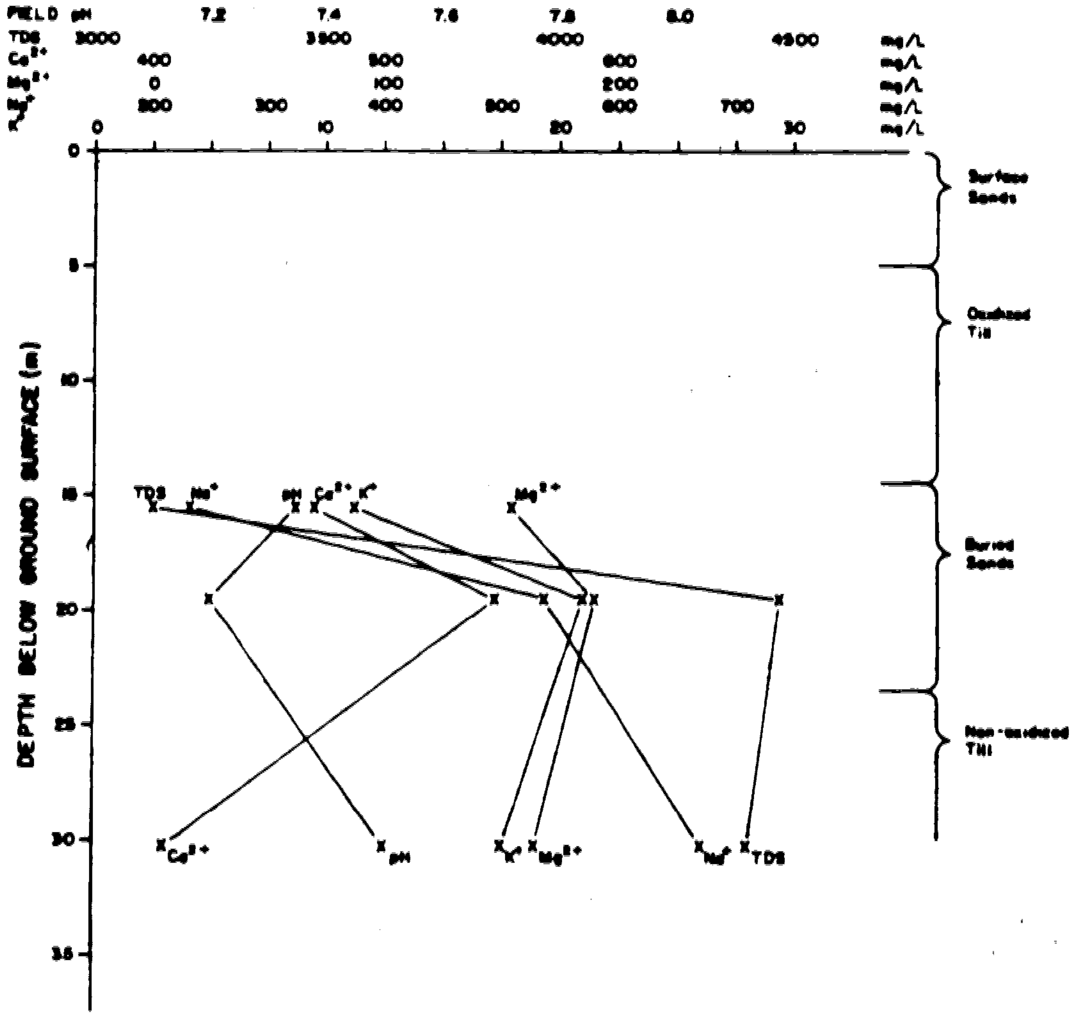


FIGURE 25a: Cation Concentration Changes in the Groundwater Beneath Piezometer Nest 1813.

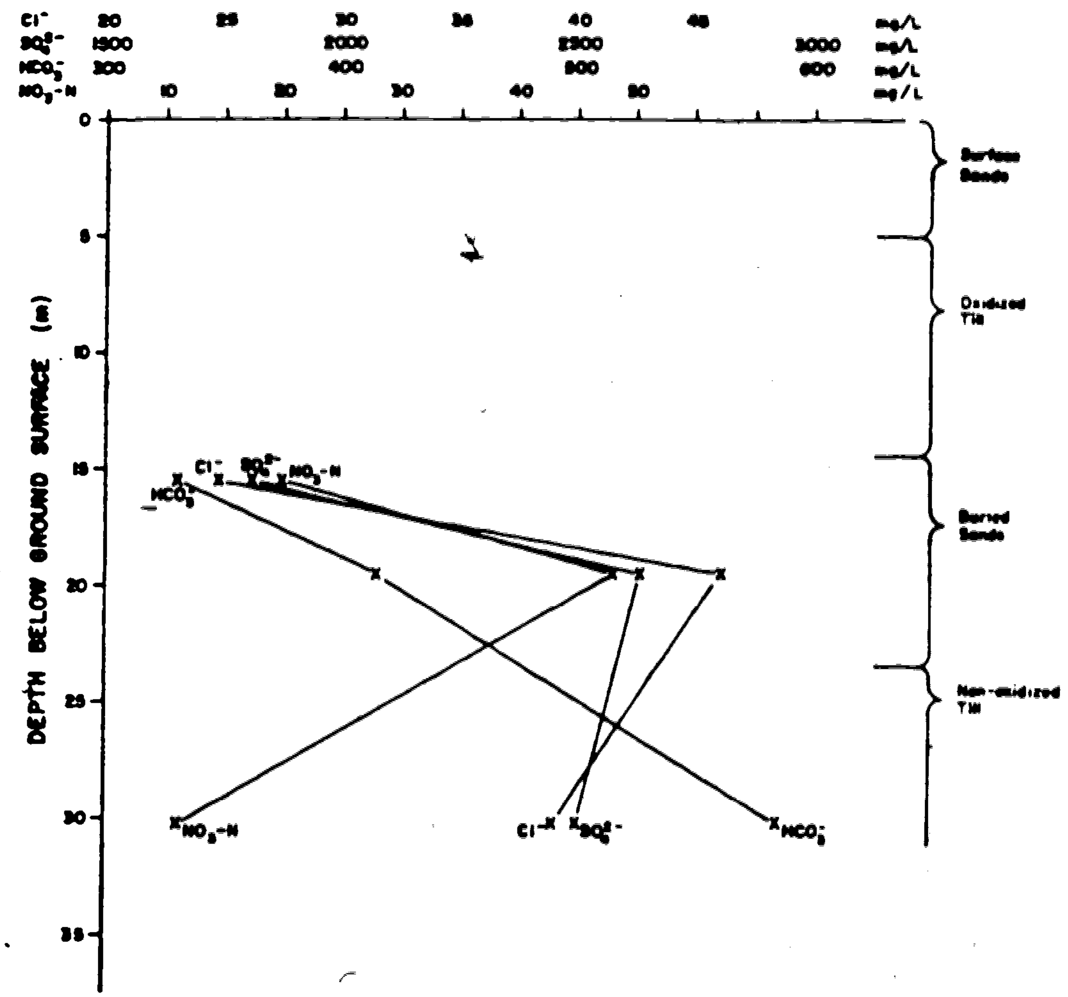


FIGURE 25b: Anion Concentration Changes in the Groundwater Beneath Piezometer Nest 1813.

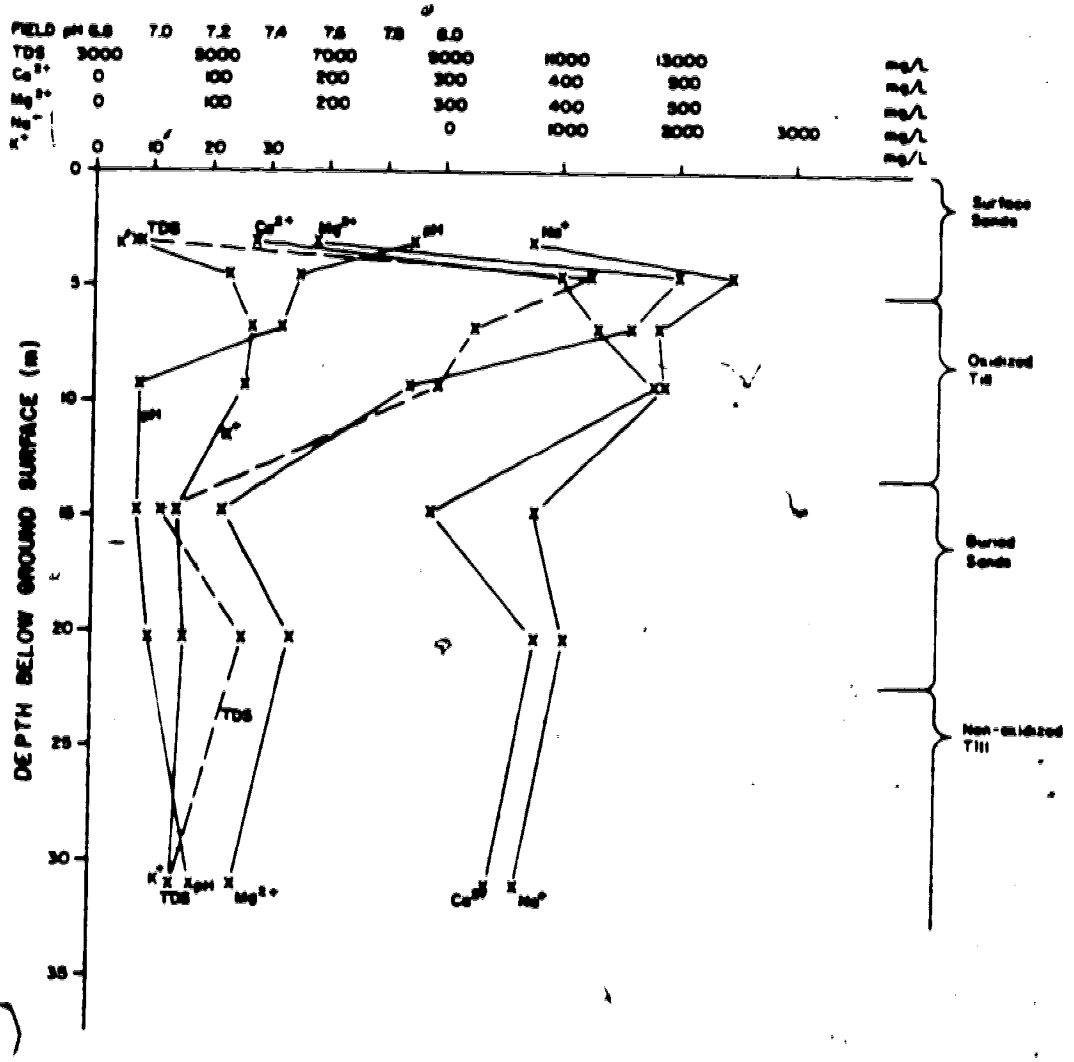


FIGURE 26a: Cation Concentration Changes in the Groundwater Beneath Piezometer Nest 1814

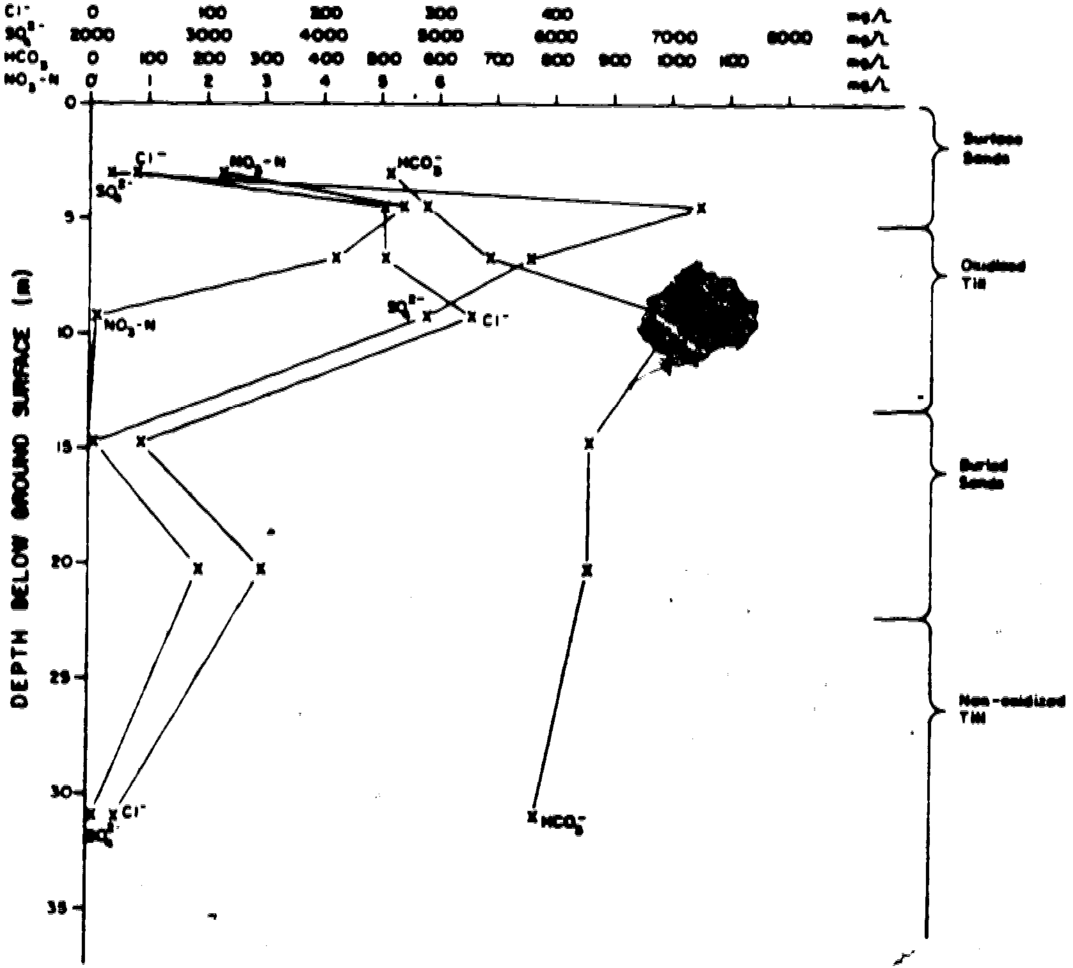


FIGURE 26b: Anion Concentration Changes in the Groundwater Beneath Piezometer Nest 1814.

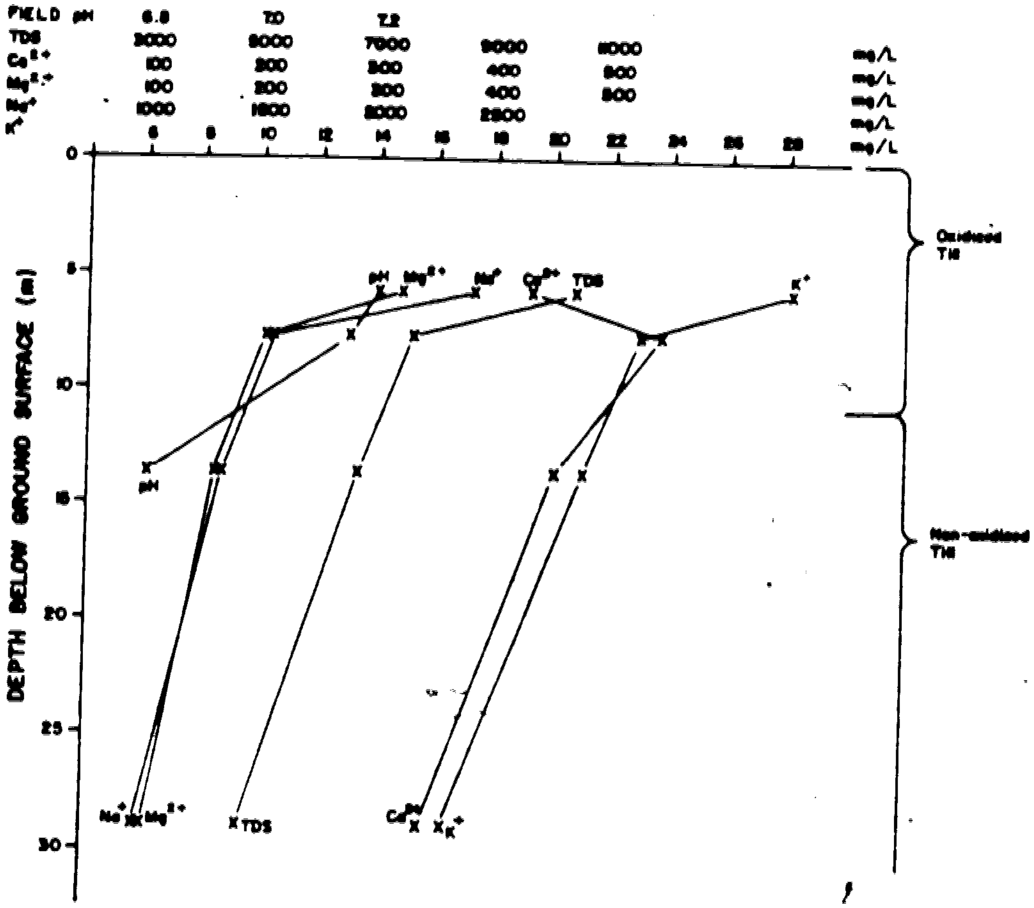


FIGURE 27a: Cation Concentration Changes in the Groundwater Beneath Piezometer Nest 1816.

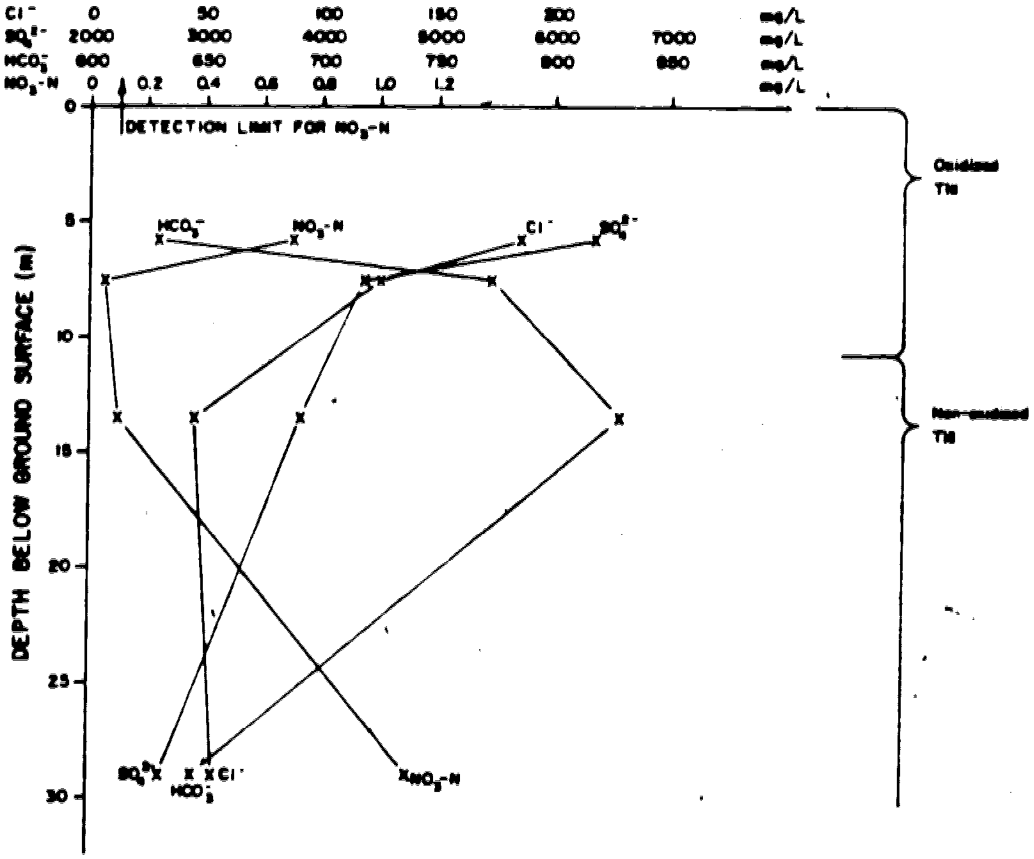


FIGURE 27b: Anion Concentration Changes in the Groundwater Beneath Piezometer Nest 1816.

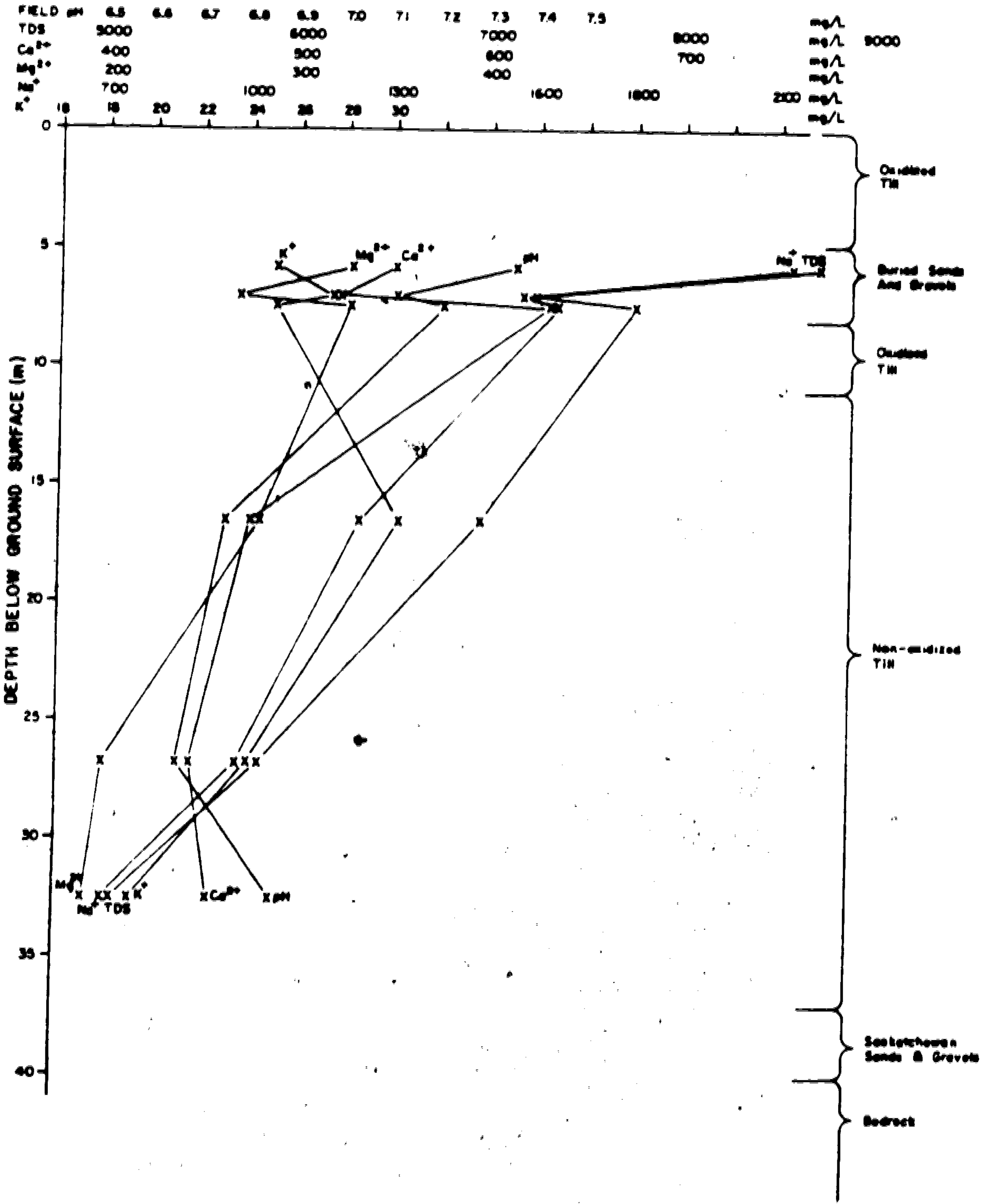


FIGURE 28a: Cation Concentration Changes in the Groundwater Beneath Piezometer Nest 1817.

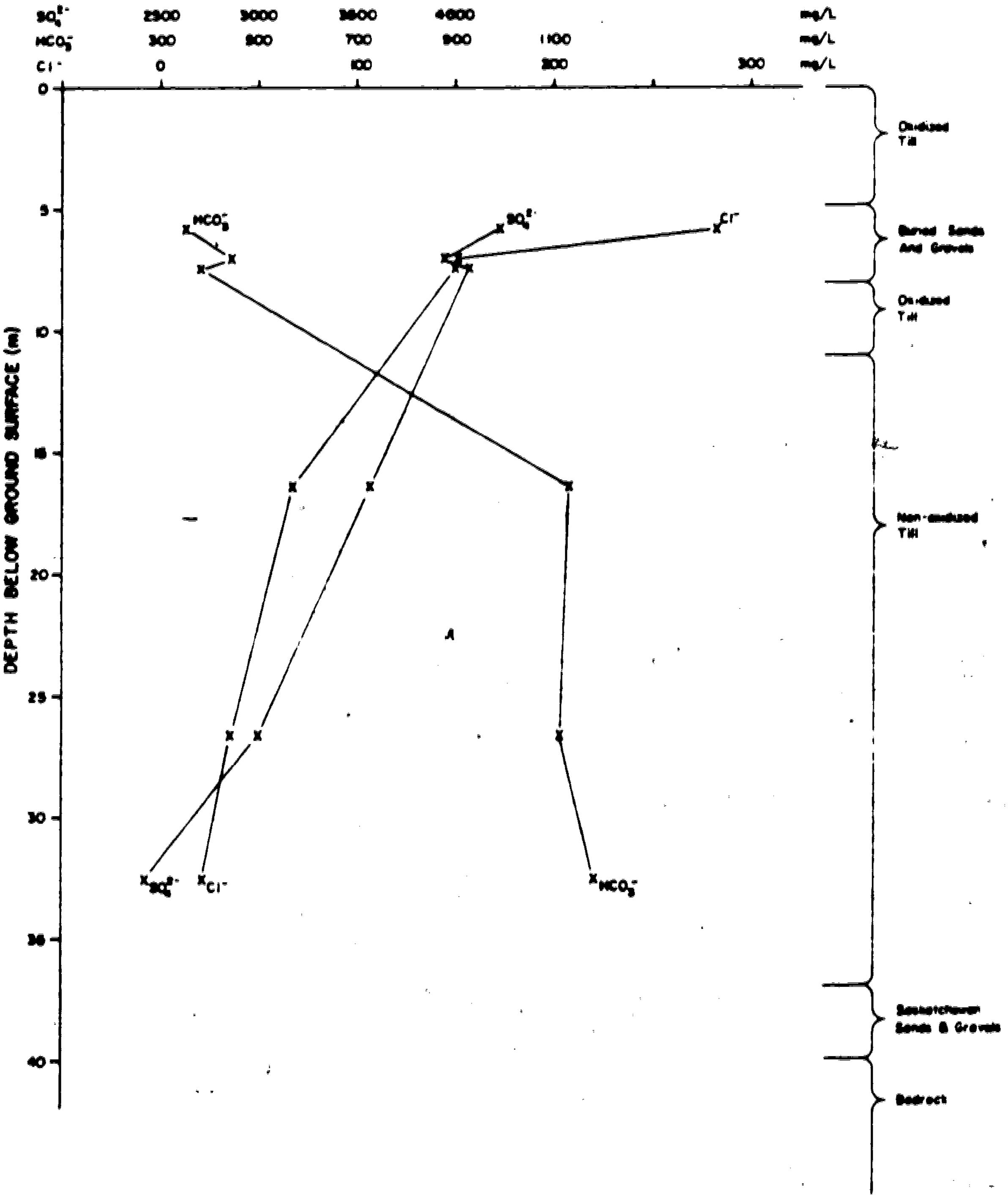


FIGURE 28b: Anion Concentration Changes in the Groundwater Beneath Piezometer Nest 1817.

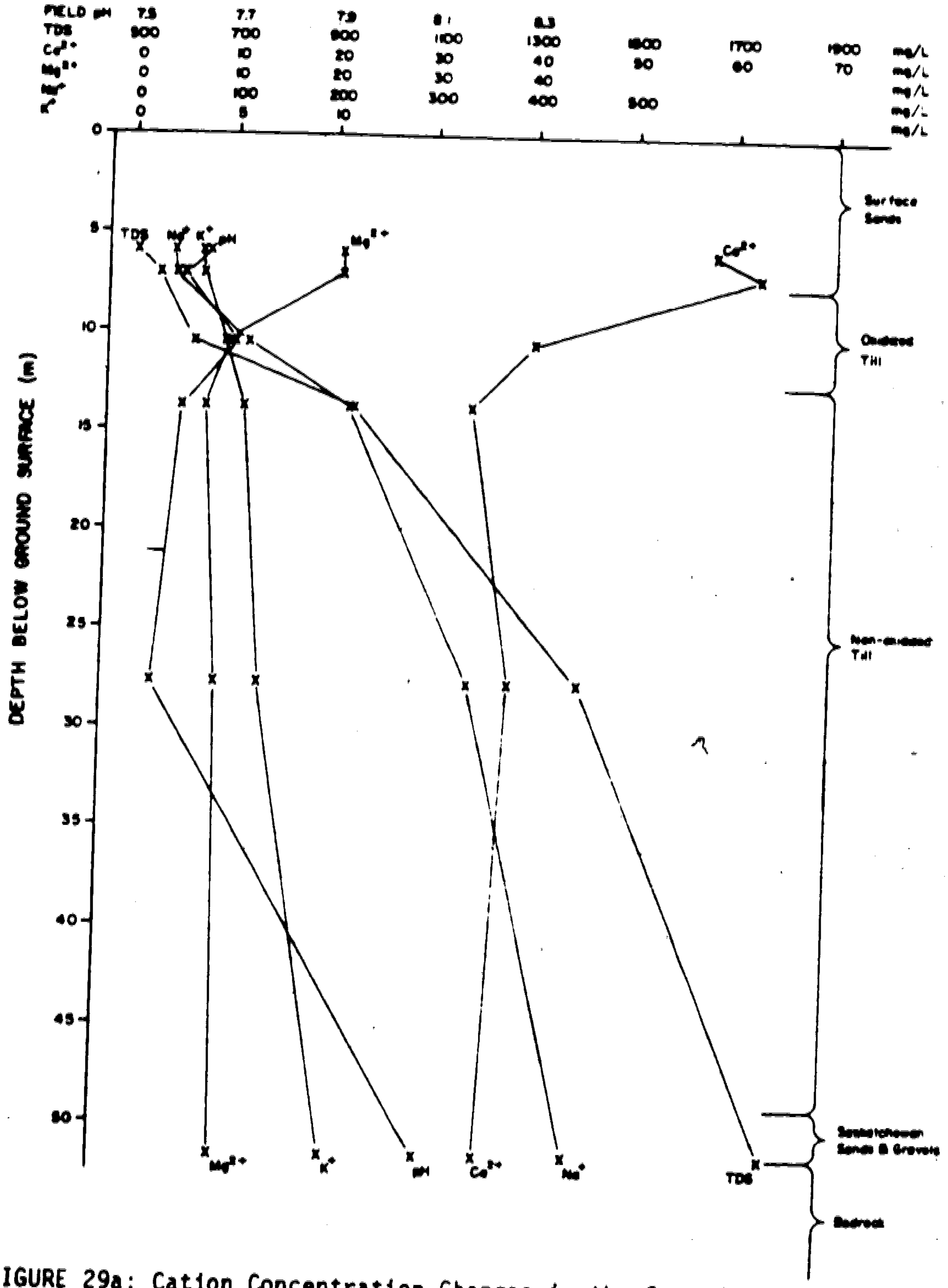


FIGURE 29a: Cation Concentration Changes in the Groundwater Beneath Piezometer Nest 1818.

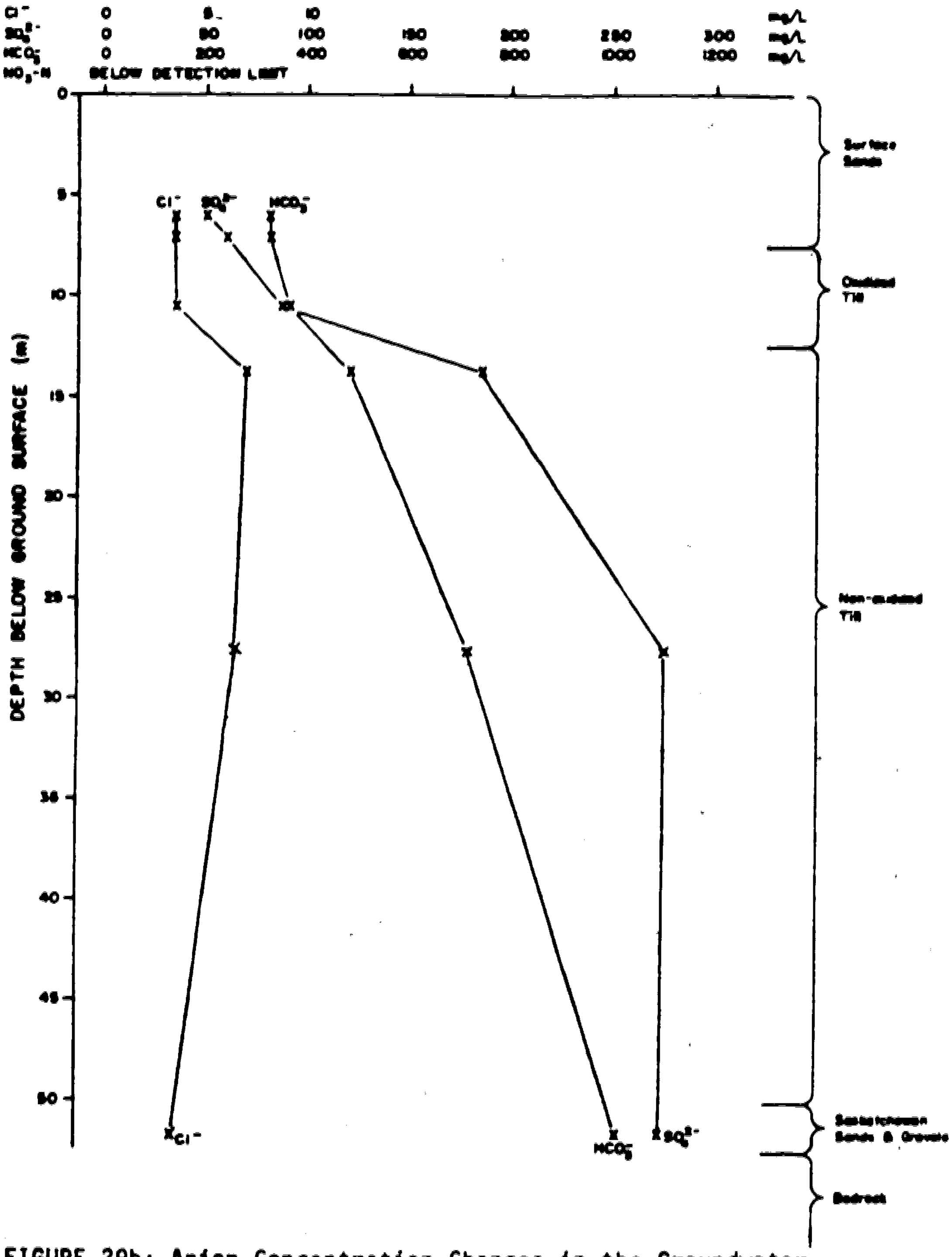


FIGURE 29b: Anion Concentration Changes in the Groundwater Beneath Piezometer Nest 1818.

Such uncertainty exists between all data points shown in figures 23 to 29. This could not be overcome because financial and time limits existed with respect to the number of piezometers which could be installed. The uncertainty which exists between data points on either side of the overburden geological boundaries is felt to be cosiderably greater than the uncertainty between data points within the same geologic unit since the varying grain size and chemical properties of the overburden deposits affect the groundwater quality differently. For this reason the plots of chemical quality of the groundwater beneath the major piezometer nests, (figures 23 to 29), will be used to illustrate changes in groundwater quality which occur within the four major overburden units. Differences between similar deposits across the study site will also be examined.

5.4.1 Groundwater Chemistry Changes Through th Surface Sands

Piezometer nests 1814 and 1818 (figures 26 and 28), each have two piezometers completed in the surface sands. The piezometers at 1814 are at 2.8m and 4.4m deep while those at 1818 are a few meters deeper at 5.9m and 7.1m. The water table at site 1814 was shallow throughout the summer of 1980 (April to September), and varied between 2.05m and 2.33m beneath the surface; at 1818 the water table was at 3.59m at the time of sampling and fluctuated only 0.1m during the summer.

At site 1814 concentrations of all species increase through the surface sands. At the base of the sands concentrations are some of the highest seen anywhere in the study site. Groundwater total dissolved solids concentration reaches 11,497mg/L (figure 26(a)), the major contributors being sodium and sulphate ions. Calcium, magnesium and bicarbonate also attain concentrations greater than 400mg/L.

Results of 5:1 extract analyses (table 15) showed that the surface sands contain some soluble salts which are dominantly carbonates and sulphates of calcium and magnesium, and sodium. Concentrations in the 5:1 aqueous sand extracts were much lower than those from the tills, with the exception of carbonate-bicarbonate, and dissolution of carbonates and sulphates by the processes already outlined cannot account for the high concentrations observed at site 1814. For example, the 6.2meq/L sodium concentration in the 5:1 extract from sample 5, taken at 3.0m beneath site 1814, represents only 142mg/L of sodium in the extract. Even assuming that not all of the sodium was removed by the 5:1 extract, this can hardly account for the sodium concentration of over 2500mg/L which was detected in the groundwater close to this point.

Chloride ion concentration at the base of the surface sands at site 1814 is high even though no source of chloride for dissolution was indicated by the 5:1 extract analysis results. Chloride is generally a very mobile yet inert anion in the flow system therefore some process seems to be concentrating the dissolved species in the shallow groundwater beneath piezometer nest 1814. Hydraulic gradients between the base of the surface sand and underlying oxidized till indicate that water movement is downward from the sands into the till. Hence upward movement of more concentrated water from the till below cannot be invoked as the cause of the high concentrations at the base of the surface sands.

Ca:Mg molar ratios for water samples from the surface sands at site 1814 (table 14), are much less than 1.0 being 0.44 and 0.49 at 2.8m and 4.4m respectively. Ca:Mg molar ratios have been discussed in an earlier section and appear to indicate that evaporation and evapotranspiration are the processes which are concentrating the soluble species. Evapotranspiration by plants removes water molecules from the unsaturated zone through selectively permeable membranes in their roots. Dissolved species are left behind in the remaining water which continues to move toward the water table and so concentrations of ionic constituents in this water become relatively enriched. Evaporation directly from the water table at site 1814 would be impeded by the very porous nature of the surface sands which would not allow capillary rise of the water to occur, direct evaporation from the water table is therefore unlikely to be the cause of the low Ca:Mg ratios in the groundwater at the water table.

Assuming the irrigation water introduced at the surface adjacent to site 1814 is pure river water with no fertilizers added, its total dissolved solids concentration would be close to that of the river, in the order of 300mg/L. To attain the observed total dissolved solids concentration of close to 11,500mg/L, approximately 97.5% of the water introduced at the surface would need to be lost through evaporation and evapotranspiration from the unsaturated zone so that only 2.5% of the water introduced at the surface reaches the water table. This represents a minimum percentage since dissolution of soluble salts will contribute some of the increase in total dissolved solids concentration. Hence, it seems likely that approximately 5% of the water added at the surface reaches the water table, (pers. comm. M.J. Hendry).

Even though evaporation and evapotranspiration effects in the unsaturated zone appear, from the Ca:Mg ratios, to be the likely cause of the high total dissolved solids concentrations at the base of the surface sands at site 1814, a problem still remains; how can the much lower total dissolved solids concentrations at the 2.8m piezometer be accounted for? The low total dissolved solids concentrations and low Ca:Mg molar ratios for samples from this piezometer are consistent through all three data sets. No reasons for these differences was identified.

In contrast with the increase in total dissolved solids with depth through the surface sands at site 1814, concentrations of most ionic species change little through the surface sands at site 1818 (figure 29). Ionic concentrations of calcium and sulphate in the groundwater increase a little with depth, suggesting that gypsum dissolution may be occurring. This is possible since the saturation indices for site 1818 (table 12) indicate undersaturation of the groundwater with respect to gypsum in the water samples from the surface sands.

✓

The constant bicarbonate ion concentration between the two piezometers in the surface sands at site 1818 (figure 29(b)) indicates that\ no net carbonate dissolution occurs in this zone. Calcite solution is probably limited by the saturation level of the water since it is supersaturated with respect to calcite by the time it reaches the 5.9m piezometer. At this depth the groundwater is just undersaturated with respect to dolomite. In order to achieve dolomite saturation, magnesium ion concentration in the groundwater must increase, however this was not observed between the two piezometers in the surface sands. Dolomite solution within the surface sands may, therefore, be supply limited.

Concentrations of all major ion species in the groundwater from the surface sands at site 1818 (figures 29(a) and 29(b)) are much lower than at site 1814 (figures 26(a) and 26(b)). Hydraulic gradients indicate that flow through the surface sands at site 1818 is downward to just above the contact with the till where the gradient stabilizes and at times is reversed. This probably indicates lateral flow within these sands just above the contact with the underlying oxidized till. 5:1 extract analysis results (table 15) show that surface sands at 1818 may have slightly lower concentrations of sodium salts and sulphates than the sands at 1814 but the difference between the deposits at the two sites is not great enough to account fo the much lower concentrations of ions in the groundwater at site 1818.

Ca:Mg molar ratios for the piezometers at 5.9m and 7.1m in the surface sands at site 1818 are greater than 1.0 (table 14), indicating that the effects of evaporation and evapotranspiration are not as pronounced as at site 1814. This is probably the reason for the differences in the chemical quality of water samples from the surface sands at these two piezometer nests and may be the result of the different vegetation types at the two sites.

In summary, the differences between ionic concentrations in the groundwater samples from the surface sands at sites 1814 and 1818 seem to be caused by differing evaporation and evapotranspiration effects within the unsaturated zone. The reasons for this are not certain but may result from different vegetation types at the two sites and their ability to remove water from the unsaturated zone.

5.4.2 Groundwater Chemistry Changes Through the Oxidized Till

Piezometer nests 1814 and 1816 each have two piezometers completed in the oxidized till unit (figures 26(a) and (b), and 27(a) and (b)). At site 1814 the piezometers are at 6.9m and 9.2m below ground surface and underlie the surface sands, while at site 1816 the piezometers in the oxidized till are at 6.1m and 7.6m beneath ground surface; the oxidized till extends to the surface.

At site 1814 (figure 26(a)) total dissolved solids concentration of the groundwater decreases between the two piezometers within the oxidized till although some ionic concentrations increase while others decrease. Calcium, sodium, bicarbonate and chloride ion concentrations in the groundwater increase and magnesium, sulphate, and nitrate-nitrogen ion concentrations decrease (figures 26(a) and (b)). pH of the groundwater also decreases between the two piezometers in the oxidized till.

The decrease in total dissolved solids concentration of the groundwater through the oxidized till is probably the result of chemical and biochemical effects on water previously affected by evapotranspiration, as it passes through the groundwater flow system. This is reflected in the Ca:Mg ratio which decreases from 0.57 at the top of the till to 1.09 at the 9.2m piezometer (table 14). The decreasing magnesium ion concentration in the groundwater accompanied by the increasing calcium concentration causes this increase in Ca:Mg molar ratio (figure 26(a)). The decrease in magnesium ion concentration may be caused by the precipitation of magnesium salts or by the removal of magnesium from the system through some other process. Precipitation of magnesium salts seems unlikely since they are usually more soluble than calcium salts and the increase in calcium and bicarbonate ion concentrations indicates that calcite is dissolving. Also, precipitation of dolomite is so sluggish that supersaturation can often exist for quite long periods. In this case, the saturation indices (table 12) indicate that the groundwater has considerably higher levels of supersaturation with respect to dolomite than those with respect to calcite because of its high magnesium ion concentration. This may induce the precipitation of dolomite in preference to calcite and will have the secondary effect of increasing calcite dissolution in order to maintain calcium and bicarbonate ion concentrations in the groundwater at equilibrium levels with respect to calcite. However, this calcite dissolution will only replace the ions lost by dolomite precipitation, hence calcite dissolution in excess of that in response to dolomite precipitation must be invoked in order to explain observed increases in calcium and bicarbonate ion concentrations in the groundwater.

The slight increase in sodium ion concentration as the groundwater moves through the oxidized till at site 1814 (figure 26(a)) may be the response to cation exchange. This could also explain part of the decrease in magnesium ion concentration. The expected decrease in calcium ion concentration is being obscured by some other process which is producing calcium ions at a greater rate than they are being adsorbed. Increase in sodium ion concentration through the oxidized till is quite small indicating that cation exchange is not occurring to the degree expected in a deposit in which 50% of the total clay minerals present is smectite (Hendry 1981). This is probably due to the confining effects of the fracture system which limit the surface area of contact between the water and the deposit and induce rapid flow velocities.

Chloride ion concentration in the groundwater increases through the oxidized till beneath site 1814 (figure 26(b)) to the highest levels observed anywhere in the study site. This may be caused by some local geologic source of chloride ions within the till although local bedrock is not known to have high chloride concentrations. The source of chloride remains obscure...

Sulphate ion concentration in the groundwater declines quite rapidly through the oxidized till at site 1814 (figure 26(b)). This decline is probably the result of attenuation by physical processes as well as the precipitation of sulphates. Gypsum precipitation could be occurring since the saturation indices indicate that the groundwater is supersaturated with respect to gypsum (table 12). The large decrease in sulphate ion concentration accompanied by the increase in bicarbonate concentration in the groundwater initially seems to indicate sulphate reduction. No hydrogen sulphide smell was detected in the samples from the oxidized till, although it may have reacted with any iron or other transition metals present to form sulphides (Bohn et al., 1979). The possibility of sulphate reduction occurring here cannot be completely ruled out.

Nitrate-nitrogen concentration decreases through the oxidized till probably because of biochemical attenuation processes such as denitrification. The loss of nitrate-nitrogen is only a few mg/L and so the secondary common ion effects of denitrification will not have any significant influence on the system.

Changes in groundwater chemistry with increasing depth through the oxidized till beneath site 1816 are somewhat different from those outlined for site 1814. At site 1816 (figure 27(a) and (b)) concentrations of all major ion species in the groundwater, except calcium and bicarbonate, decrease through the oxidized till. The uppermost piezometer in the oxidized tills at site 1816 is located just beneath the water table, therefore evaporation and evapotranspiration effects which occur in the unsaturated zone may account for the high ionic concentrations of major ions in the water samples taken from this piezometer, (TDS=10,351mg/L) in a similar way to those described for site 1814. This is also reflected in the Ca:Mg molar ratio at this piezometer of 0.8 (table 14). The 5:1 extract analysis results showed that the oxidized till contains many soluble salts (table 15, figures 20(a) to 20(j)) therefore it is surprising that the concentrations of major ion species in the groundwater decrease rather than increasing with depth through the oxidized till. The occurrence of some physical attenuation process is suggested by the observed decreasing concentration of mobile ions in the groundwater with increasing depth, such as chloride ion concentration. Precipitation of supersaturated species may also account for some of the observed decrease in ionic concentrations.

The anomalous increase in calcium and bicarbonate ion concentrations as the groundwater moves downward through the oxidized till deposits at site 1816 (figure 27(a) and (b)) may be caused by calcite and/or gypsum dissolution. Saturation levels with respect to calcite (table 12) increase with depth through the oxidized till whereas dolomite and gypsum saturation levels are constant. The constant dolomite saturation level suggests that the decrease in magnesium ion concentration through the oxidized till at this site is balanced by the increase in calcium and bicarbonate ion concentrations. Calcite solution is more likely to be occurring than gypsum solution since the bicarbonate ion concentration in the groundwater and its saturation level with respect to calcite increase, whereas sulphate ion concentration decreases and gypsum saturation level remains constant. Sulphate reduction may also be responsible for decreased sulphate ion concentration and increased bicarbonate ion concentration. No smell of hydrogen sulphide was detected within the oxidized till at site 1816, although this does not totally preclude the possibility of sulphate reduction. Positive redox potentials were also detected within these tills so that sulphate reduction seems unlikely.

The decreasing pH through the oxidized till indicates that some process is producing hydrogen ions at a faster rate than they are being used up for calcite dissolution. The two major processes which have been seen to produce hydrogen ions are carbon dioxide dissolution and pyrite oxidation, but these occur dominantly in the unsaturated zone. The process which is producing hydrogen ions within the oxidized till is therefore unidentified.

The observed changes in the chemical quality of groundwater as it moves through the oxidized till beneath sites 1814 and 1816 are similar with respect to some ionic species but vary with respect to others. These changes represent the net effect of processes and process interactions which occur within the oxidized till. No definite conclusions can be drawn as to which processes are occurring since the relative magnitudes of their effects are unknown. However, some speculation on the processes which are causing the observed changes in water quality has been made after considering the limited information availible here.

5.4.3 Groundwater Chemistry Changes Through the Buried Sands

Three piezometer nests have piezometers completed in the buried sand deposits. Sites 1813 (figure 25(a) and (b)) and 1814 (figure 26 (a) and (b)) each have two piezometers in a large buried sand lens which is continuous between the two sites (figures 5 and 6) and has lateral flow within it from north to south. There are three piezometers in the discontinuous buried sand and gravel lens at site 1817 (figure 28(a) and (b)).

At site 1813 (figure 25(a) and (b)), the upper piezometer is completed close to the top of the buried sand lens at a depth of 15.4m while the lower piezometer in the sands is in the middle of the lens at 19.5m. The water table at this site is much deeper than in other parts of the study site, at approximately 14m beneath the surface located within the oxidized till just above the sand lens.

Concentrations of all major ion species in the groundwater increase with depth through the buried sand lens beneath site 1813 (figure 25(a) and (b)), in the direction of vertical flow, while pH decreases. This increase in ionic concentrations is probably the net result of soluble salt dissolution within the sands .

The ions present in the highest concentrations in the water samples from the upper part of the sand lens are calcium and sulphate, which were probably derived by the solution of gypsum through the great depth of unsaturated oxidized till above. Sodium and sulphate concentrations in the groundwater show the most marked increase through the buried sands. These may have been derived by the solution of sodium sulphate precipitates which Grossman (1968) identified in buried channel deposits of the Interior Plains region of Canada. When the observed increase in sodium and sulphate ion concentrations through the buried sands are converted to comparible units, the increase in sulphate ion concentration is found to be in excess of that which can be attributed to sodium sulphate dissolution, indicating that some additional source of sulphate exists within the buried sands.

Part of the increase in calcium and sulphate ion concentrations as the groundwater moves through the buried sands may be the result of gypsum dissolution within the sands, since groundwater saturation levels with respect to gypsum increase slightly through the deposit. Alternatively, and probably more likely, the observed increase in saturation level with respect to gypsum may be the result of the common ion effect through contribution of calcium by calcite and dolomite, and sulphate from sodium sulphate.

Increased calcium, magnesium and bicarbonate ion concentrations in the groundwater as it moves through the buried sands, probably result from the dissolution of calcite and dolomite. The slight decrease in saturation levels with respect to these minerals is caused by decreased pH which increases the ability of the groundwater to dissolve calcite and dolomite.

At piezometer nest 1814 (figure 26(a) and (b)), two piezometers are completed in the buried sands at 14.8m and 20.3m below ground surface. The saturated thickness of deposits above the buried sands is much greater at site 1814 than at 1813 because the water table here is only a little over 2m beneath ground surface.

Concentrations of all major ion species in the groundwater, as well as its pH, increase through the buried sand deposits beneath site 1814. Nitrate concentrations are below detection limit (0.1mg/L) at this site. There is an obvious similarity in the concentration trends through the sands at sites 1813 and 1814. The same processes which were described for site 1813 will also apply to this site.

Although similar general trends occur in ionic concentration variations in the groundwater as it moves through the buried sand deposits at sites 1813 and 1814, the absolute concentrations of ions are often quite different. In order to illustrate these differences, figure 30 shows the groundwater quality changes through the buried sand deposits beneath each of the sites, plotted on the same scales. The boundaries of the buried sands are located at the depths they occur at site 1814.

Total dissolved solids concentration in the groundwater samples is lower at site 1813 than at site 1814, mostly as a consequence of lower sodium and bicarbonate ion concentrations at site 1813. Sulphate and chloride concentrations in the groundwater are also lower at site

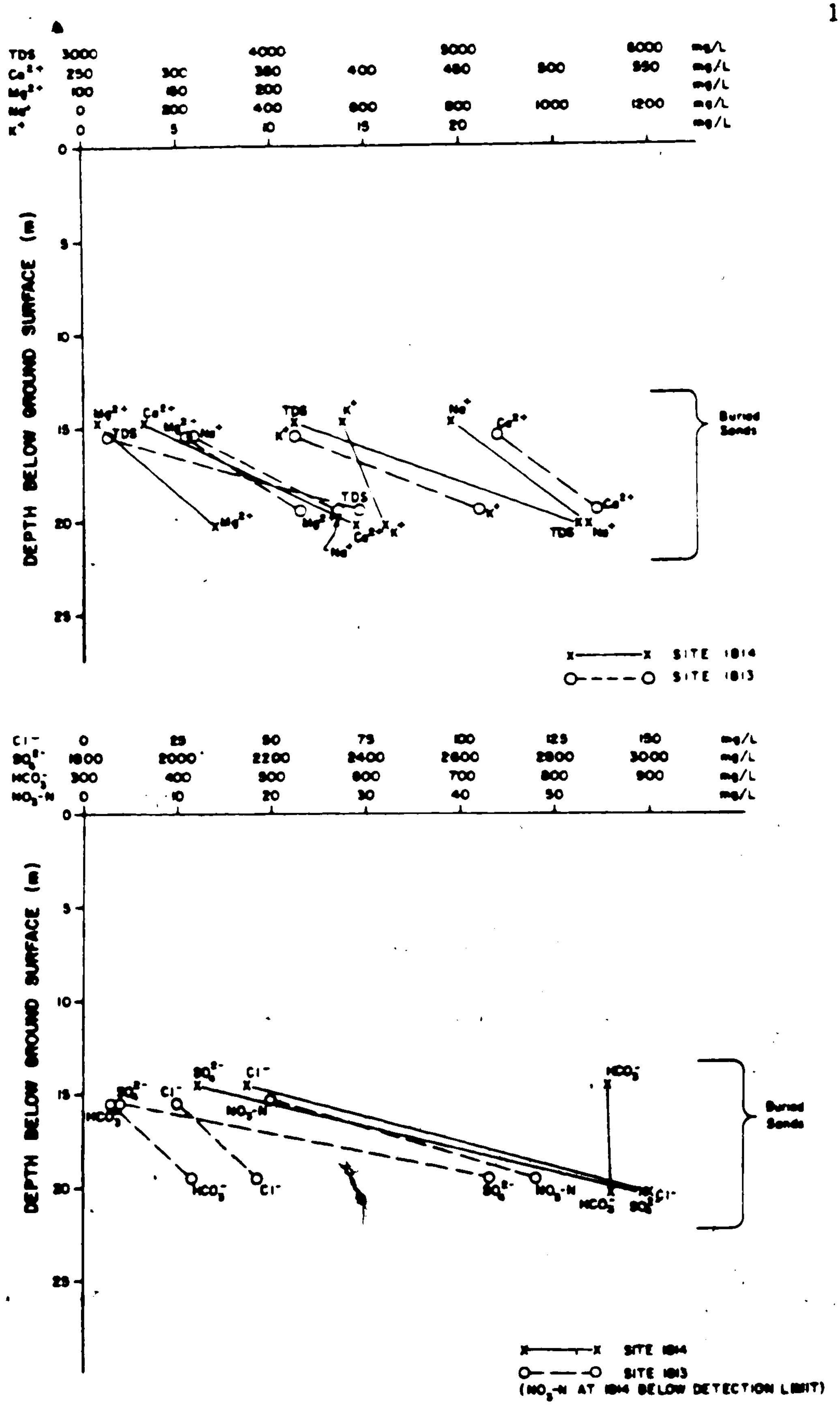


FIGURE 30: Changes in Groundwater Chemistry Through the Buried Sands.

1813. Groundwater calcium, magnesium, potassium and nitrate-nitrogen ion concentrations are higher at site 1813 than 1814.

The observed changes in water quality between sites 1814 and 1813 in the direction of the lateral flow component are a net result of the processes occuring within the sand lens accompanied by the input of water between site 1814 and 1813 both from the oxidized till above and from the underlying non-oxidized till. The input of water from below is indicated by the negative (upward) gradients that exist between the buried sands and underlying non-oxidized till at sites 1814 and 1813, although the upward gradients are considerably stronger at 1813 than at 1814, (Alberta Agriculture, 1981). Since no information is availible at present concerning the amount and quality of the water which enters the buried sands at successive points between the two sites, the exact processes which occur within the sands cannot be explained. However, it has already been seen that water from the non-oxidized till generally has lower ionic concentrations than the water in the sands so that the input of water from below may explain the decreasing ionic concentrations from site 1814 to 1813, at least at the base of the sand lens.

The difference in water table elevation between site 1814 and 1813 may be responsible for some of the observed differences in groundwater quality between the upper two piezometers at these sites. Above site 1814 the zone of unsaturated flow is very small but its thickness gradually increases toward site 1813. This may affect the quality of the water which is entering the buried sand deposit from the oxidized till above. The exact effect of this difference in saturated zone thickness cannot be assessed because processes occurring in the unsaturated zone are complex. Since obtaining water samples from the unsaturated zone is so difficult, it has not been included in the scope of this study.

It is interesting to note that nitrate-nitrogen concentrations increase between sites 1814 and 1813. This must indicate that some nitrate-nitrogen source is present between the two piezometer nests. An irrigated field exists at the surface between sites 1814 and 1813 and the observed increase in groundwater nitrate-nitrogen concentration between them may be the result of excess fertilizers which have found their way to the groundwater and into the buried sands through the fracture network in the oxidized till. Alternatively, some geologic source of nitrate-nitrogen may exist within the buried sand deposits. Whichever is the case, nitrate-nitrogen concentrations at the base of the sand lens are greater than the safe potable limit (10mg/L). Unfortunately no piezometers exist which allow samples to be collected from further south in the sand lens along the flow path to see whether nitrate-nitrogen concentrations continue to increase or whether they decrease again by natural attenuation or denitrification processes. This should be checked if any further research is undertaken in the area. The water table distribution across the site (figure 8) suggests that water flowing south in this buried channel deposit, intersects an east-west trending channel and then flows either east or west to the river. A few samples of river water were taken during the summer of 1980 downstream of where this buried sand deposit would intersect the river, nitrate-nitrogen levels in the river water were found to be below detection limits. Nitrate pollution of the Oldman River at this point along its channel is therefore not a problem at present.

Three piezometers are located in the discontinuous buried sand and gravel lens beneath site 1817 at depths of 5.7m, 7.0m, and 7.4m (figure 28(a) and(b)). The water table at this site ranged between 4.95m and 5.18m beneath ground surface during the field season.

The general trend within the buried sands at site 1817 is for ionic concentrations in the groundwater to decrease between 5.7m and 7.0m, and then increase again toward 7.4m. Potassium and bicarbonate ion concentrations showed the opposite pattern.

Dissolution of soluble salts, enhanced by carbon dioxide dissolution and pyrite oxidation, probably occur in the unsaturated zone within the tills above the buried sands and gravels accompanied by the effects of evaporation and evapotranspiration on percolating water. Therefore, once the water reaches the saturated zone it has already attained moderately high ionic concentrations. Total dissolved solids concentration is 8692mg/L at the shallowest piezometer in this sand lens.

Brief examination of the data from the second and third data sets (tables 5 and 6) shows that relative increases and decreases in concentrations of various ions in the groundwater within the buried sands and gravels at site 1817, were inconsistent through the summer of 1980. For example in the results from the second sample set, calcium and magnesium ion concentrations decrease consistently through the lens while sodium concentration increased consistently. This is different to th[illegible]uation shown in figure 28 for the first sample set. Th[illegible] variability through the summer, and the variable concentrations with depth through the lens may be the result of rapid hydraulic connection between the surface and the sand lens through the fracture network in the oxidized till, which causes variable dilution within the lens in response to discontinuous recharge events. Hydraulic gradients within this sand lens also vary temporally, maybe for the same reason.

Processes within these buried sands and gravels at site 1817 probably include the dissolution of some soluble salts since ionic concentrations in the groundwater increase between the lower two piezometers, however, the variable concentrations of major ion species precludes anything more than pure speculation on the processes which may be occurring here.

5.4.4 Groundwater Chemistry Changes Through the Non-oxidized Till

Four piezometer nests have more than one piezometer completed in the non-oxidized till. These are site 1811 (figure 23), 1816 (figure 27(a) and (b)), 1817 (figure 28(a) and(b)), and 1818 (figure 29(a) and (b)). There are general similarities between sites 1811 and 1818 where concentrations of most ionic species in the groundwater increase through the non-oxidized till, and sites 1816 and 1817 where concentrations decrease through the till. Concentrations of all major ion species in the groundwater are much lower at site 1811 and 1818 than at site 1816 and 1817. Possible reasons for this have already been outlined.

All three piezometers at site 1811 are located in the non-oxidized till. Therefore by the time the groundwater reaches the upper piezometer at this site it already has a characteristic chemistry which results from the dissolution of soluble species and other modifying chemical and biochemical processes occurring within the oxidized till above.

The dominant species which cause the general increase in total dissolved solids as the groundwater moves through the non-oxidized till beneath site 1811 are sodium and bicarbonate (figure 23). The major part of the increase in total dissolved solids concentration occurs through the lower part of the non-oxidized till. Conversely, most of the decrease in calcium and magnesium ion concentration also occurs through the lower portion of the profile. Sulphate ion concentration varies very little while nitrate-nitrogen concentration is consistently below detection limit and pH increases with depth.

The observed changes in calcium, magnesium, sodium, and bicarbonate ion concentrations in the groundwater suggest that cation exchange is occurring as the groundwater moves down through the non-oxidized tills. Calcium and magnesium are being adsorbed onto the exchange sites in return for sodium, causing decreased calcium and magnesium concentrations accompanied by an increase in sodium in the groundwater.

Increased bicarbonate ion concentration and pH, as seen in an earlier section, also result from cation exchange because carbonate dissolution occurs in response to the loss of calcium and magnesium ions from the groundwater. The increase in bicarbonate ion concentration observed between the upper and the lower piezometer could therefore be explained by this process.

Other processes which have already been discussed and are known to increase bicarbonate ion concentrations in groundwater, are denitrification and sulphate reduction. Nitrate concentrations in the groundwater beneath piezometer nest 1811 are consistently below detection limit so that denitrification is in no way indicated. Sulphate concentration decreases in the upper part of the profile whereas bicarbonate ion concentration increases in the lower portion so that no direct link can be made between sulphate reduction and increasing bicarbonate ion concentration either.

The observed changes in groundwater chemical quality beneath site 1811 are complex and are the end result of an interaction of processes, probably including cation exchange. The limited data availible here make it impossible to account for the observed changes in detail, only speculation of the processes occurring is possible.

Piezometer nest 1818 (figure 29(a) and (b)) has two piezometers completed in the non-oxidized till. At this site ionic concentrations of all species in the groundwater increase or remain constant as the groundwater moves through the non-oxidized till. Dissolution of soluble salts is probably the major process which accounts for the observed concentration increase at site 1818 (figure 29(a) and(b)). No evidence exists for modification of groundwater chemistry by processes such as cation exchange or sulphate reduction although this is not to say that they do not occur.

The reasons for the much lower groundwater concentrations at sites 1811 and 1818 than at other sites across the study site have already been examined. Further research is necessary to understand fully why these differences exist across the site. Detailed sampling of the deposits at sites 1811 and 1818 would be useful so that chemical, grain size and biological analyses of the deposits could be made. Also, installation of more extensive piezometer nests would be helpful.

Site 1816 (figure 27(a) and (b)) has two piezometers completed in the non-oxidized till. Concentrations of all major ion species in the groundwater, except nitrate-nitrogen and chloride, decrease as the groundwater moves downward through the till. Nitrate and chloride concentrations increase very slightly.

When the upper piezometer in the non-oxidized till at site 1816 was sampled, it had the distinct odour of hydrogen sulphide. The redox potentials at this site were -135mv at both piezometers in the non-oxidized till. The occurrence of sulphate reduction in the non-oxidized till indicated by this odour can explain the decreasing sulphate concentrations as the groundwater moves through the deposit and the occurrence of the maximum bicarbonate ion concentration within the profile. The lack of hydrogen sulphide odour at the lower piezometer and the decrease in bicarbonate concentration between the two piezometers may indicate that sulphide reduction occurs dominantly in the vicinity of the upper piezometer in the non-oxidized till although this is not necessarily the case since the hydrogen sulphide may be consumed in the manner described earlier. The decrease in other major ion concentrations through the profile is caused by physical attenuation mechanisms through the matrix of the non-oxidized till and, in the case of calcium and magnesium at least, by precipitation of carbonates induced by the sulphate reduction process.

At site 1817 (figure 28(a) and (b)) three piezometers are completed in the non-oxidized till, and in a similar way to site 1816, concentrations generally decrease through this unit. The smell of hydrogen sulphide was detected at the lower two piezometers at this site throughout the field season. The redox potentials were -25mv and -110mv at 26.7m and 32.5m respectively, therefore sulphate reduction is also occurring at this site.

The decreasing calcium ion concentration in the groundwater and decreasing pH, which usually accompany sulphate reduction are not observed in the groundwater from the non-oxidized till beneath site 1817. Some other process or combination of processes which produces calcium and depletes hydrogen ion concentrations in the groundwater must be dominating the effects of sulphate reduction so that they are not observed as changes in groundwater quality between piezometers.

There is no evidence of sulphate reduction at the upper piezometer in the non-oxidized till at site 1817, therefore it cannot be used to account for the particularly high bicarbonate ion concentration at this piezometer. However, a plot of change in nitrate-nitrogen concentration through this profile (figure 31) shows that nitrate-nitrogen concentrations decrease markedly through the upper portion of the non-oxidized till so that high bicarbonate ion concentrations can be attributed to denitrification.

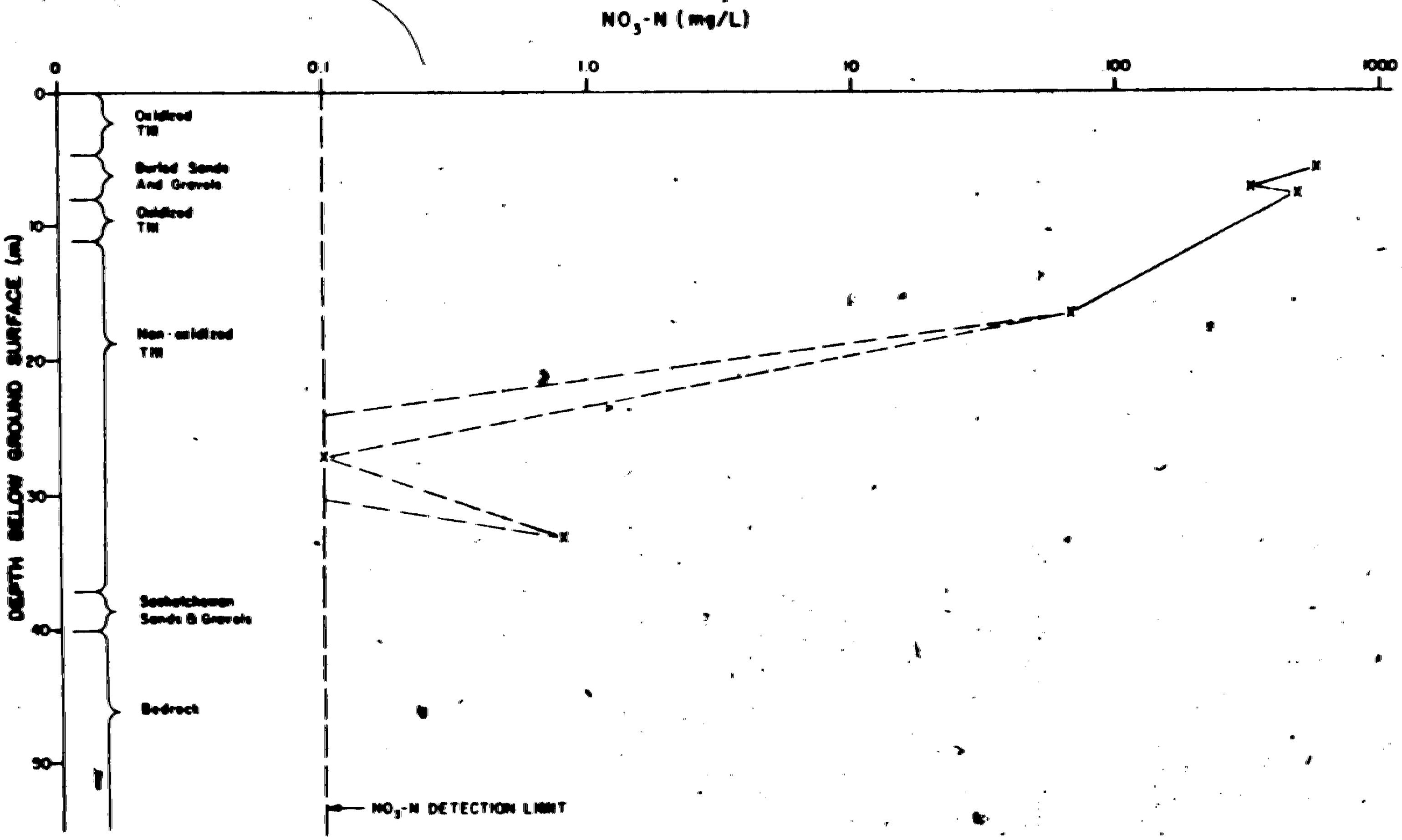


FIGURE 31: Changes in Nitrate-Nitrogen Concentration in the Groundwater Beneath Piezometer Nest 1817.

Sodium concentrations are high in the groundwater samples extracted from the non-oxidized till beneath site 1817 (figure 28(a)), this would therefore be an ideal site for reverse cation exchange. This could be the case in the lower part of the profile since sodium concentration in the groundwater decreases more rapidly than at shallower depths and this is accompanied by calcium ion concentration increase. Therefore the observed changes are the net result of sulphate reduction and perhaps some reverse cation exchange.

In summary, the results of groundwater sample analyses from piezometer nests across the study site have indicated that the processes described at the beginning of this chapter are all occurring in places beneath the study site. There is often a lack of similarity when changes in groundwater chemistry through the same geologic unit are examined at more than one piezometer nest. This is the result of the interaction of processes within the flow system. When the quality of a water sample is analysed, only the end result of these process interactions is observed. Without knowing the relative magnitude of process and effect it is impossible to state with certainty the combination of processes which occurred to produce the observed response. The best that can be done is to estimate the processes which appear to be occurring, having considered the physical conditions existing in the system.

The next section will briefly summarize the sequence of processes which is felt to produce the changes in groundwater quality observed through a generalized profile beneath the study site. This is a best estimate from th data availible.

5.5 Summary of the Groundwater Hydrochemical Evolu

Water enters the soil horizons from precipita snowmelt or irrigation. Precipitation and snowmel negligible total dissolved solids concentrations water used for irrigating has concentrations in t 200mg/L to 300mg/L. Fertilizers may be added to th wa although the amounts and types are unknown. The semi-arid climate, means that much of the water is returned to the atmosphere through evaporation and evapotranspiration instead of reaching the water table, however some water continues to percolate down through the unsaturated zone. Burnett (1981) estimates that 10cm of the water introduced at the surface reaches the water table each year.

In this unsaturated zone carbon dioxide partial pressures are high due to the respiration of micro-organisms and microbial decay, and carbon dioxide dissolves in the water resulting in a pH decrease. Pyrite oxidation may also help to decrease pH. As the pH of the water decreases it becomes more agressive with respect to carbonate dissolution and begins to dissolve calcite and dolomite. Gypsum, which was formerly precipitated in the unsaturated zone by the joint effects of pyrite oxidation, carbonate dissolution and evaporation/evapotranspiration, is also dissolved.

By the time the water reaches the water table it already has quite high concentrations of calcium, magnesium, sodium, bicarbonate and sulphate ions, as well as some potassium and chloride. Sodium cations and sulphate anions are usually the dominant species. Saturation indices with respect to calcite, dolomite and gypsum indicate levels close to or above saturation at or just beneath the water table.

As the groundwater moves downward, through the overburden deposits concentrations of ionic species change because of physical, chemical and biochemical attenuation processes which affect the groundwater chemistry. Total dissolved solids concentrations just below the water table are often quite high due to processes occurring in the unsaturated zone. The groundwater then begins to flow down through the fracture network and matrix of the oxidized till. Since the fractures have a very low porosity, flow velocities are much more rapid through the fractures than through the higher porosity matrix. The rapid movement of water in the fractures probably limits the processes which alter the groundwater quality in the oxidized till. Some processes must be occurring in the oxidized till since total dissolved solids concentrations change through this unit, often decreasing from the very high concentrations present in the groundwater from the surface sands. This is somewhat surprising since the 5:1 aqueous extracts showed that there are many soluble salts within the oxidized till matrix which could be dissolved by the groundwater; however, rapid flow velocities and supersaturation of the groundwater with respect to calcite, dolomite, and gypsum, limit the dissolution of soluble species. The decrease in total dissolved solids which is often observed through the oxidized till is probably due to the precipitation of these supersaturated species. This may also explain the high concentrations of soluble salts found in the 5:1 extracts from the oxidized till matrix.

Cation exchange was expected to occur in the oxidized till since the till is rich in smectite; however, little direct evidence of cation exchange is observed, such as increasing sodium concentration accompanied by increasing bicarbonate ion concentration and pH. Once again this is probably the result of the very rapid flow velocities in the fractures. Further research into the relative concentrations of ionic species in the matrix water and fracture water would be desireable to determine the processes which occur in the oxidized till unit, and the degree of exchange of soluble species which occurs between the groundwater in the fractures and the till matrix.

Groundwater continues to move vertically downward from the oxidized tills into the buried sand deposits where ionic concentrations of most species at the top of the buried sands are lower than in the oxidized till. Concentrations increase again through the buried sands probably because of dissolution of soluble salts, and concentrations recover, sometimes reaching the levels observed within the oxidized till. Lateral flow occurs in the major buried sand deposits, and ionic species concentrations decrease in the direction of lateral flow. This concentration decrease appears to result from differences in the quality of water entering the buried sand deposit along the lateral flow path, both from the oxidized till above and the non-oxidized till below.

Once the groundwater enters the non-oxidized tills, concentrations of ionic species decrease with depth. This is the result of physical, chemical and biochemical attenuation mechanisms and the precipitation of soluble salts. Cation exchange can occur since residence times are longer and areas of surface contact greater than in the oxidized till. Denitrification and sulphate reduction are also possible due to the low redox potentials and presence of necessary bacteria within these deposits. This enhances the precipitation of carbonates.

By the time the water reaches the lower parts of the overburden deposits at the top of the Saskatchewan sands and gravels, total dissolved solids concentrations have generally been reduced to between 2000mg/L and 4000mg/L.

5.5.1 Effects of Irrigation Return Flow

In the groundwater flow diagrams (figures 9 to 11) presented in section 2.5, groundwater was seen to enter the Saskatchewan sands and gravels and then begin to move laterally toward the Oldman River. Therefore the water which is entering the Saskatchewan sands and gravels from the overburden deposits above at concentrations between 2000mg/L and 4000mg/L is also expected to be reaching the Oldman River.

River water samples taken from downstream of the study site have total dissolved solids concentrations in the range 200mg/L to 300mg/L which are not significantly different from those taken upstream of the site. Therefore, at present the irrigation return flow does not appear to be adversely affecting the quality of the Oldman River. This may be the result of further attenuation processes occurring within the Saskatchewan sands and gravels which cannot be identified here because of the lack of piezometers completed in this unit. Alternatively, it may be the result of dilution of the return flow water to concentrations beyond detection when it enters the Oldman River.

Natural attenuation processes which occur within the overburden deposits, especially the non-oxidized till are very important in this groundwater flow system. These processes reduce the generally high ionic concentrations present in the shallow groundwater which are the result of evaporation and evapotranspiration effects in the unsaturated zone as well as dissolution of soluble salts near the top of the oxidized till.

No potential future pollution of the Oldman River by irrigation return flows from the study site can be foreseen from interpretation of the data available at present. Natural physical, chemical and biochemical processes within the overburden deposits are keeping potential problems under control.

6. Summary and Conclusions

The major findings of this study are summarized here;

(1)The study site, which is situated in southern Alberta, comprises a hummocky plain underlain by up to forty meters of laterally and vertically variable Laurentide glacial deposits.

(2)The general sequence of overburden deposits from surface to bedrock is;

1. Surface outwash sand
2. Oxidized fractured till
3. Buried outwash sand
4. Non-oxidized unfractured till
5. Saskatchewan sands and gravels
6. Bedrock

(3)Mean hydraulic conductivities for these units vary over four orders of magnitude. The surface sands have the greatest mean conductivity (1.3×10^{-3}cm/sec), while mean conductivity in the buried sands is 1.1×10^{-4}cm/sec. The oxidized and non-oxidized till have mean conductivities of 7.1×10^{-4}cm/sec and 5.9×10^{-7}cm/sec respectively. Conductivity of the non-oxidized till decreases slightly with depth, from 7.9×10^{-7}cm/sec in the upper parts to 5.0×10^{-7}cm/sec deeper in the unit. Hydraulic conductivity of the locally unsaturated Saskatchewan sands and gravels was found from one single well response test to be 6.5×10^{-3}cm/sec.

(4)The elevation of the water table varies little across the study site except for a depression in the southeast which is the result of drainage through a buried channel sand deposit. The water table is generally 2 to 5 meters beneath the surface.

(5)Flow direction through the glacial overburden deposits is dominantly vertically downward. Water entering the large buried sand lens beneath piezometer nests 1813 and 1814, flows south toward the edge of the buried preglacial Oldman River Valley and then flows either west or east toward the river,

(6)Flow velocities in the overburden deposits are generally quite rapid since they are induced by low porosities. Fracture flow in the oxidized till is especially rapid since the fracture porosity is very low ($2x10^{-3}$) and flow velocities may reach $6x10^{-1}$m/day.

(7)5:1 aqueous extracts from bulk samples of surface sands and oxidized till show that the till matrix contains much greater concentrations of water soluble salts than the sands.

(8)Groundwater chemical quality within the saturated zone of the overburden deposits is highly variable. Concentrations of all ionic species are significantly lower in water samples from the deposits beneath sites 1811 and 1818 than those from across the remainder of the study site. The reasons for this remain unclear.

(9)Groundwater field pH is consistently lower than laboratory pH. This probably results from degassing of dissolved carbon dioxide during transportation, storage and analysis of water samples.

(10)Results of dissolved oxygen concentration and redox potential measurements performed in the field show no correlation with depth of sample or with each other. Measuring these variables with the water sample in contact with the atmosphere probably affects the samples. A flow cell technique would have been desireable.

(11)Partial pressures of carbon dioxide with which the water samples were in equilibrium are all greater than atmospheric pCO_2. Processes occurring in the overburden deposits which produce carbon dioxide, such as respiration of micro-organisms, denitrification and sulphate reduction are probably responsible for this.

(12)Ca:Mg molar ratios in the groundwater are often less than one in the shallow saturated zones, caused by evaporation and evapotranspiration effects on the groundwater in the unsaturated zone above. Ca:Mg molar ratios increase with depth through the sequence of overburden deposits.

(13)The highest total dissolved solids concentration in the groundwater from beneath the study site is found in samples from just beneath the water table. This is probably due to the concentrating effects of evaporation and evapotranspiration in the unsaturated zone above.

(14)The combined effects of pyrite oxidation, carbon dioxide dissolution, calcite and dolomite dissolution and evapotranspiration in the unsaturated zone lead to the precipitation of gypsum during periods of water deficit. This is subsequently dissolved by percolation water during recharge events.

(15)Saturation of the groundwater with respect to calcite, dolomite and gypsum is attained just beneath the water table, indicating the presence of soluble minerals in the unsaturated zone.

(16)Sodium and sulphate are the ions present in the highest concentrations in the groundwater over most of the study site. Other major ions present in the groundwater are calcium, magnesium and bicarbonate. Nitrate-nitrogen and chloride are locally present in high concentrations.

(17)Concentrations of most major ion species present in the groundwater decrease through the overburden deposits by physical, chemical, and biochemical attenuation processes such as precipitation, cation exchange, denitrification and sulphate reduction.

(18)Local increases in the concentrations of ionic species along the groundwater flow path sometimes occur, such as by the dissolution of soluble salts through the buried sands. Bicarbonate ion concentrations in the groundwater increase through the non-oxidized till since bicarbonate ions are produced by denitrification and sulphate reduction.

(19)Cation exchange and the dissolution of soluble salts does not occur to the degree expected in the oxidized tills. This is probably caused by the rapid flow velocities through the fractures.

(20)By the time the groundwater reaches the Saskatchewan sands and gravels, total dissolved solids concentrations are generally in the order of 2000mg/L to 4000mg/L, this is significantly lower than the 11500mg/L concentrations observed just beneath the water table.

(21)Nitrate-nitrogen concentrations in the groundwater, which are often a problem in irrigation return flows, do not pose a contamination threat to the Oldman River at this site. Locally, nitrate concentrations reach 500mg/L in buried sand deposits and this nitrate may have been derived from the tills above, however, natural denitrification processes within the nonoxidized tills rapidly reduce these nitrate concentrations below detection limit.

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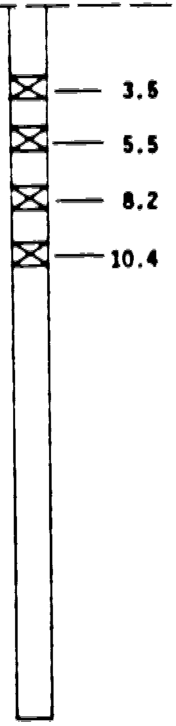
Appendix; Hydrogeologic Borehole Logs

HYDROGEOLOGIC LOG

BOREHOLE No. 1809-E-1
Page 1 of 1

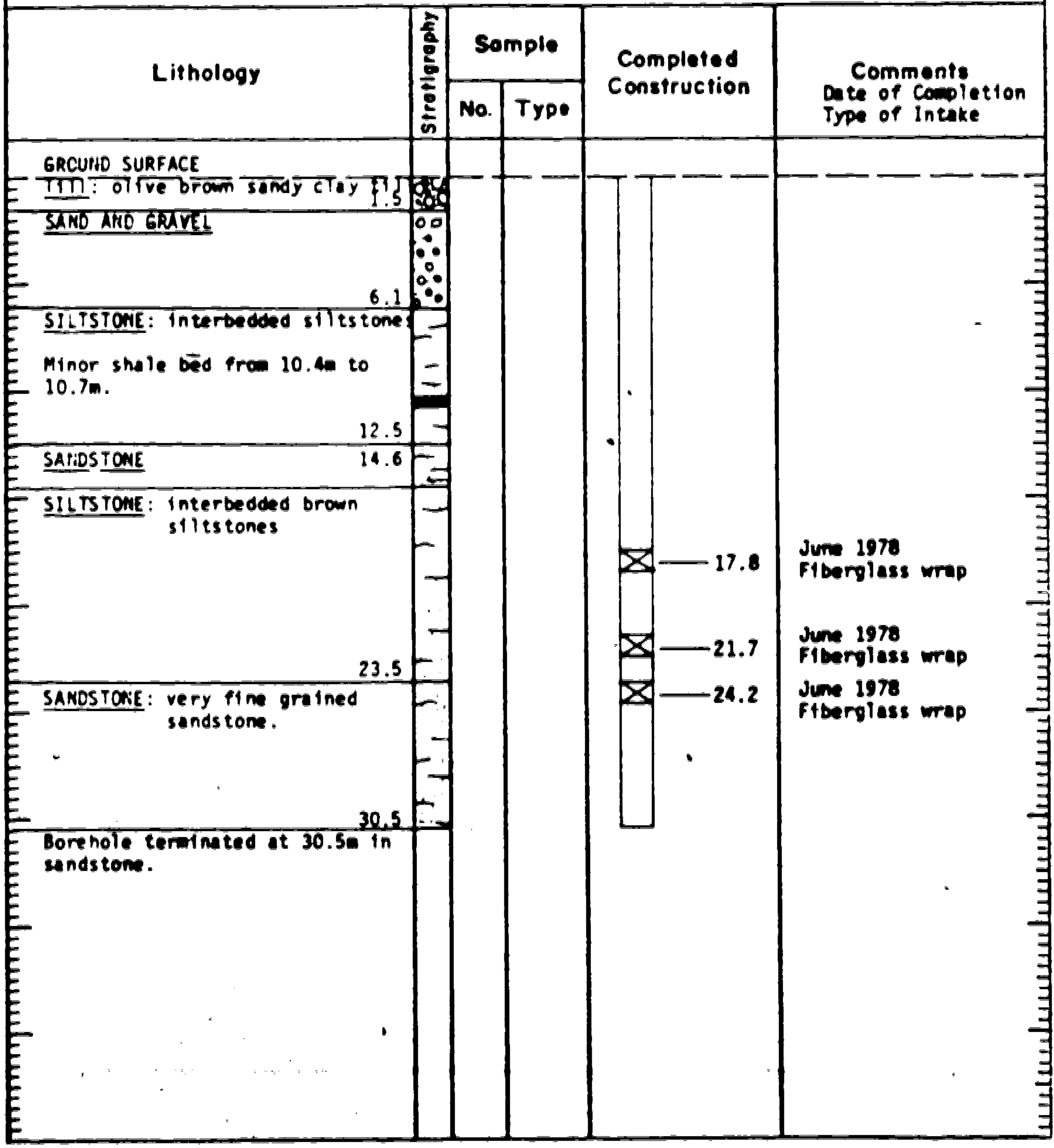
Project TARER IRRIGATION RETURN FLOW
Reference elevation 732.8
Type of drilling Rotary
Contractor Alta. Environment
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig F.A. 1000
Logged by M.C., et al
Drill fluid Water
Checked by
Date drilled June 29, 1978
Purpose of hole Stratigraphic control and ground water observation

Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
GROUND SURFACE					
TILL: olive brown sandy clay till, becoming c. grained at 2.4m. 4.3				 3.5	April 1979 Fiberglass wrap
Gravel: c. sand to fine gravel, primarily quartz grains. 6.6				5.5	April 1979 Fiberglass wrap
SILTSTONE: siltstone interbedded with brown shales. Shales vary in hardness from hard to soft and are often carbonaceous. Bentonitic sandstone from 6.6 to 7.1m. Thin soft coal layers at 6.6m, 7.1m and 12.8m. 30.5				8.2 10.4	April 1979 Fiberglass wrap April 1979 Fiberglass wrap
Borehole terminated at 30.5m in siltstone.					

HYDROGEOLOGIC LOG

BOREHOLE No 1010-E-1
Page 1 of 1

Project TABER IRRIGATION RETURN FLOW
Reference elevation 733.1
Type of drilling Rotary
Contractor Alta. Environment
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig FA 1000
Logged by M.C., et al.
Drill fluid Water
Checked by
Date drilled June 29, 1978
Purpose of hole Stratigraphic control and ground water observation.



GROUND SURFACE

TILL: olive brown sandy clay fil 1.5

SAND AND GRAVEL 6.1

SILTSTONE: interbedded siltstones

Minor shale bed from 10.4m to 10.7m. 12.5

SANDSTONE 14.6

SILTSTONE: interbedded brown siltstones 23.5

SANDSTONE: very fine grained sandstone. 30.5

Borehole terminated at 30.5m in sandstone.

17.8 June 1978 Fiberglass wrap

21.7 June 1978 Fiberglass wrap

24.2 June 1978 Fiberglass wrap

HYDROGEOLOGIC LOG

BOREHOLE No. 1811-E-1
Page 1 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 764.6
Type of drilling Rotary
Contractor Alta. Environment
Rig FA 1000
Logged by M.C., et al.
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Drill fluid Water
Checked by
Date drilled June 30, 1978
Purpose of hole Stratigraphic control and ground water observation.

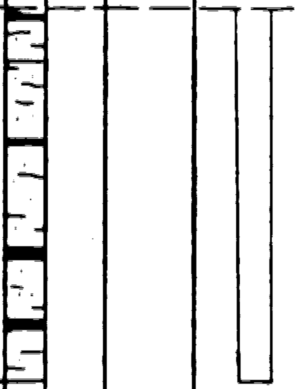
Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
GROUND SURFACE					
SAND AND GRAVEL: c. sand and gravel. 5.8					
TILL: dark grayish brown c. sandy clay till becoming a denser dark olive grey f. sandy clay till at 7.0m. Thin (0.2m) gravel layer at 11.3m. 17.0					
SAND AND GRAVEL 18.0					
TILL: olive gray c. sandy clay till. 23.2				17.4	March 1979 Well screen
				22.1	March 1979 Well screen
SAND AND GRAVEL: c. sand to f. gravel, primarily quartz and granitic grains, occasional coal chips. 27.7				24.5	March 1979 Well screen
SILTSTONE: siltstone with thin interbeds of brown shale throughout. Major shale beds (1.0 m and 0.3m thick) at 32.0m (containing carbonaceous and coal layers) and 55.8m.					

HYDROGEOLOGIC LOG

BOREHOLE No. 1811-E-1
Page 2 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 764.6
Type of drilling Rotary
Contractor Alta. Environment
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig FA 1000
Logged by M.C., et al
Drill fluid Water
Checked by
Date drilled June 30, 1978
Purpose of hole Stratigraphic control and ground water observation.

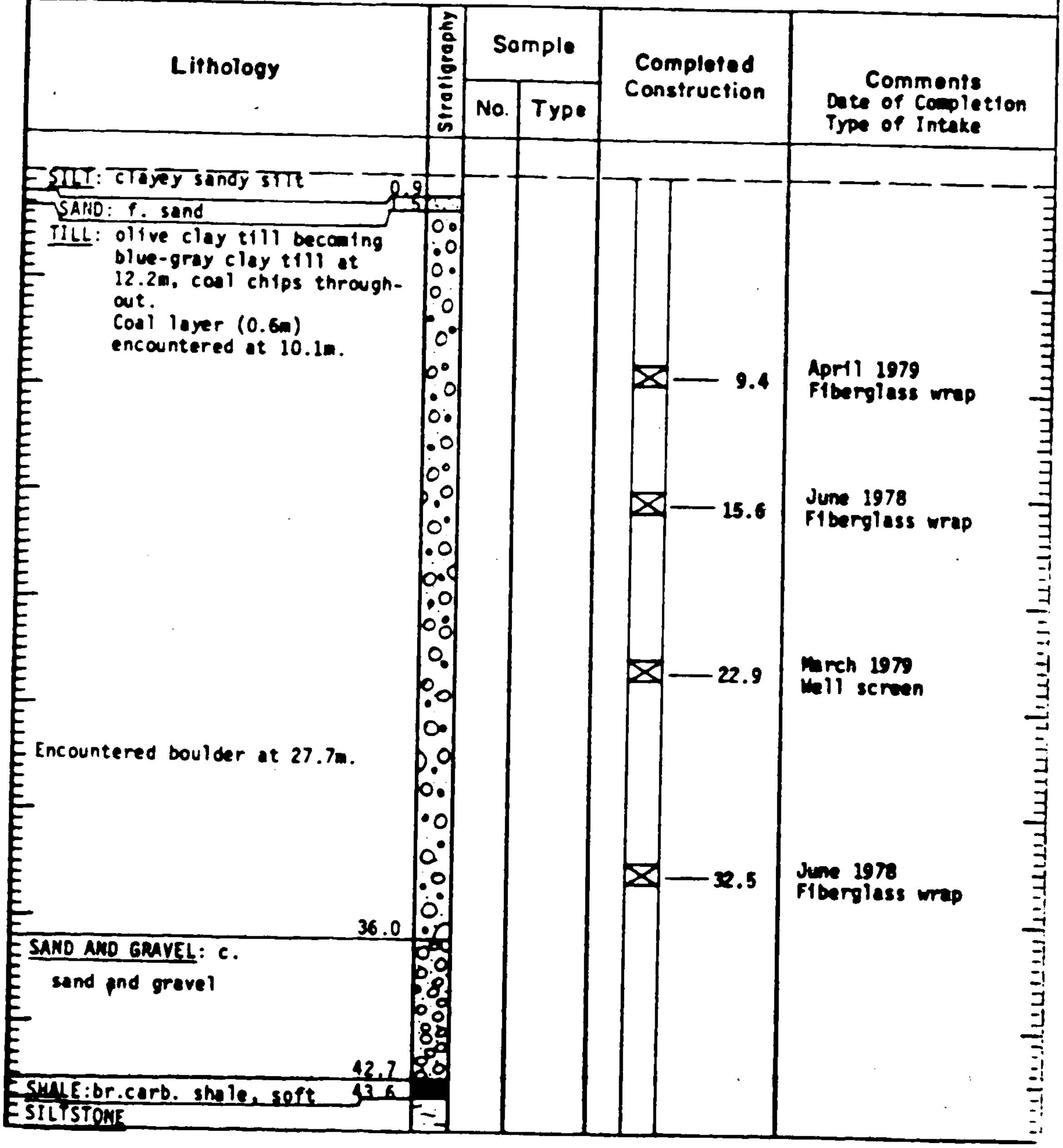
Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments
45 SILTSTONE: continued Thin coal layer at 45.1m. Hard sandstone bed (0.3m) at 55.8m. 61.0					
Borehole terminated at 61.0m in siltstone.					



HYDROGEOLOGIC LOG

BOREHOLE No 1812-E-1
Page 1 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 776.1
Type of drilling Rotary
Contractor Alta. Environment
Rig FA 1000
Logged by M.C. et al
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Drill fluid Water
Checked by
Date drilled June 30, 1978
Purpose of hole Stratigraphic control and ground water observation.

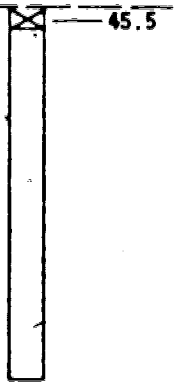


Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
SILT: clayey sandy silt 0.9					
SAND: f. sand 1.5					
TILL: olive clay till becoming blue-gray clay till at 12.2m, coal chips throughout. Coal layer (0.6m) encountered at 10.1m.				9.4	April 1979 Fiberglass wrap
				15.6	June 1978 Fiberglass wrap
				22.9	March 1979 Well screen
Encountered boulder at 27.7m.				32.5	June 1978 Fiberglass wrap
36.0					
SAND AND GRAVEL: c. sand and gravel 42.7					
SHALE: br.carb. shale, soft 43.6					
SILTSTONE					

HYDROGEOLOGIC LOG

BOREHOLE No. 1812-E-1
Page 2 of 2

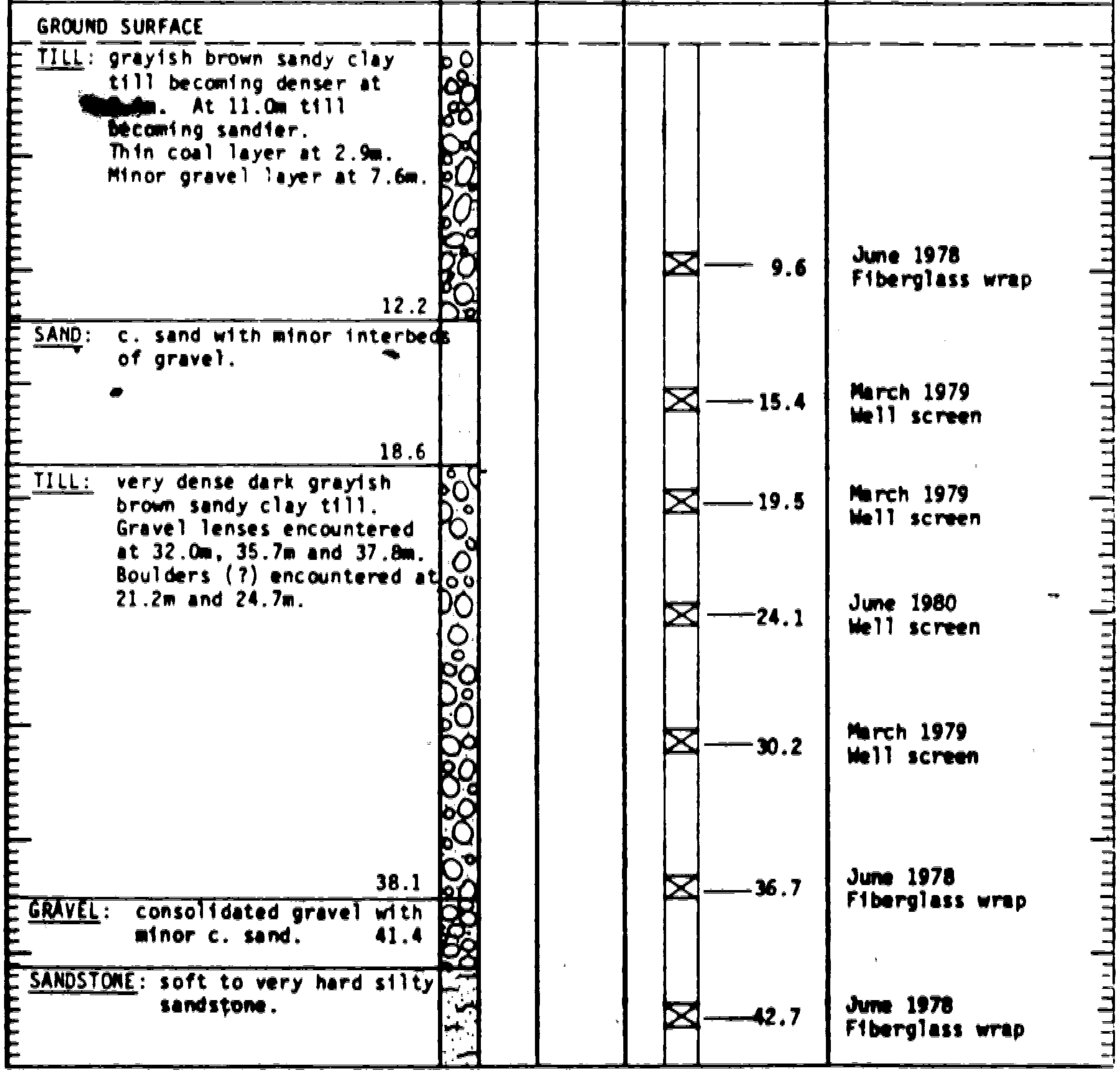
Project TABER IRRIGATION RETURN FLOW.
Reference elevation 776.1
Type of drilling Rotary
Contractor Alta. Environment
Elevation type · Surveyed ☒ Altimeter ☐ From map ☐
Rig FA 1000
Logged by M.C., et al
Drill fluid Water
Checked by
Date drilled June 30, 1978
Purpose of hole Stratigraphic control and ground water observation.

Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
45					
SILTSTONE: continued c. siltstone thin carb. shale at 46.9m 49.4				 45.5	June 1978 Fiberglass wrap
SANDSTONE: f. sandstone, soft. Thin, brown siltstone at 51.5m. Thin, hard shale at 57.6m. 61.0					
Borehole terminated at 61.0m in sandstone.					

HYDROGEOLOGIC LOG

BOREHOLE No 1813-E-1
Page 1 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 776.2
Type of drilling Rotary
Contractor Alta. Environment
Rig FA 1000
Logged by M.C., et al
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Drill fluid Water
Checked by
Date drilled July 1, 1978
Purpose of hole Stratigraphic control and ground water observation.



Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
GROUND SURFACE					
TILL: grayish brown sandy clay till becoming denser at [redacted]. At 11.0m till becoming sandier. Thin coal layer at 2.9m. Minor gravel layer at 7.6m.				9.6	June 1978 Fiberglass wrap
12.2					
SAND: c. sand with minor interbeds of gravel.				15.4	March 1979 Well screen
18.6					
TILL: very dense dark grayish brown sandy clay till. Gravel lenses encountered at 32.0m, 35.7m and 37.8m. Boulders (?) encountered at 21.2m and 24.7m.				19.5	March 1979 Well screen
				24.1	June 1980 Well screen
				30.2	March 1979 Well screen
38.1				36.7	June 1978 Fiberglass wrap
GRAVEL: consolidated gravel with minor c. sand. 41.4					
SANDSTONE: soft to very hard silty sandstone.				42.7	June 1978 Fiberglass wrap

HYDROGEOLOGIC LOG

BOREHOLE No. 1813-E-1
Page 2 of 2

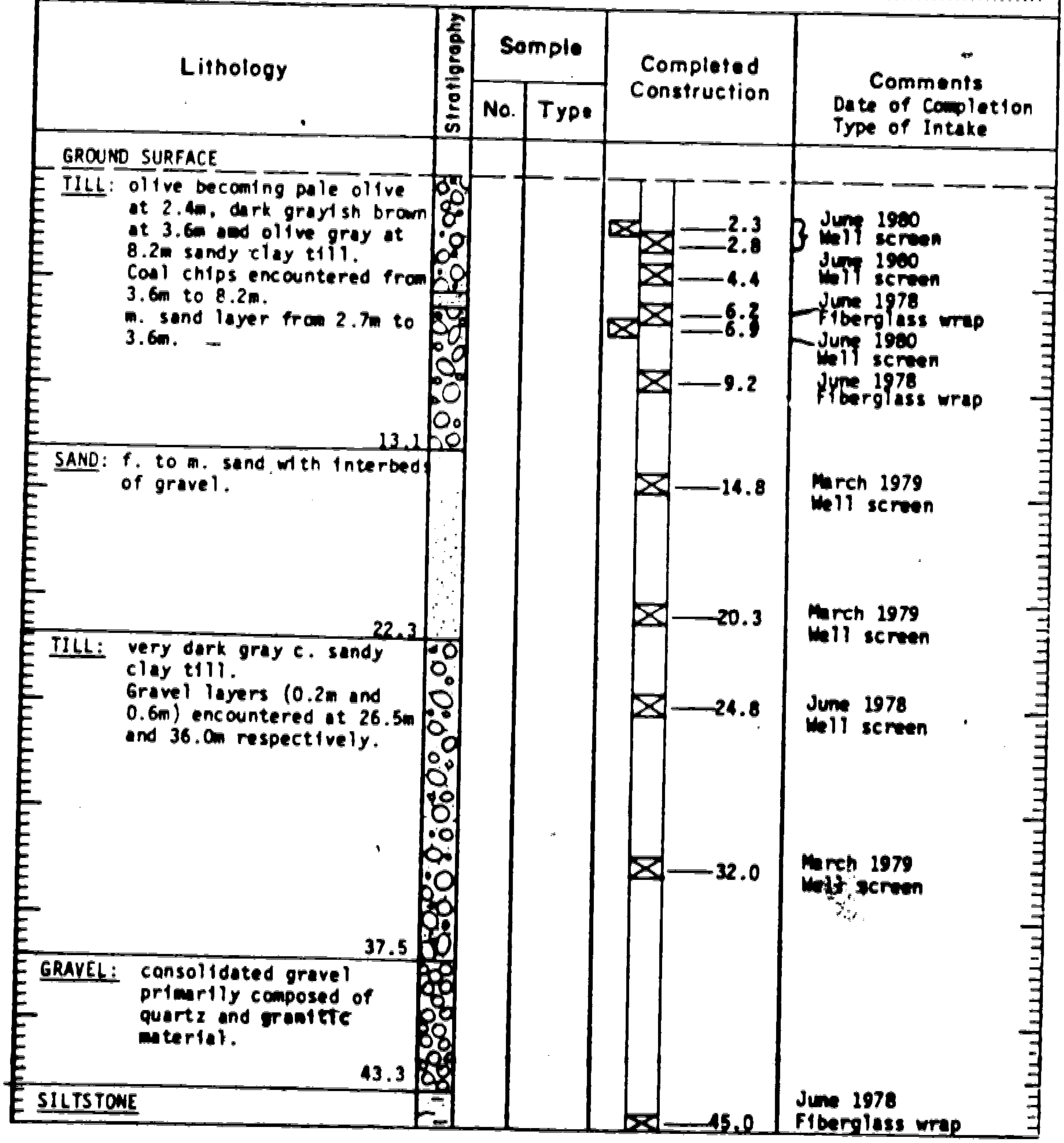
Project TABER IRRIGATION RETURN FLOW
Reference elevation 776.2
Type of drilling Rotary
Contractor Alta. Environment
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig FA 1000
Logged by M.C., et al
Drill fluid Water
Checked by
Date drilled July 1, 1978
Purpose of hole Stratigraphic control and ground water observation.

Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments
45					
SANDSTONE: continued Coal seam (0.3m) at 45.4m. Bentonitic sandstone from 45.4 m to 50.9m. Shale interbeds from 51.5m to 54.5m. 54.5					
SILTSTONE 58.2					
SANDSTONE: c. sandstone 61.0					
Borehole terminated at 61.0m in sandstone.					

HYDROGEOLOGIC LOG

BOREHOLE No. 1814-E-1
Page 1 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 773.8
Type of drilling Rotary
Contractor Alta. Environment
Rig FA 1000
Logged by M.C., et al
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Drill fluid Water
Checked by
Date drilled July 1, 1978
Purpose of hole Stratigraphic control and ground water observation.



Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake

GROUND SURFACE

TILL: olive becoming pale olive at 2.4m, dark grayish brown at 3.6m and olive gray at 8.2m sandy clay till. Coal chips encountered from 3.6m to 8.2m. m. sand layer from 2.7m to 3.6m.

13.1

SAND: f. to m. sand with interbeds of gravel.

22.3

TILL: very dark gray c. sandy clay till. Gravel layers (0.2m and 0.6m) encountered at 26.5m and 36.0m respectively.

37.5

GRAVEL: consolidated gravel primarily composed of quartz and granitic material.

43.3

SILTSTONE

Completed Construction	Comments
2.3	June 1980 Well screen
2.8	
4.4	June 1980 Well screen
6.2	June 1978 Fiberglass wrap
6.9	June 1980 Well screen
9.2	June 1978 Fiberglass wrap
14.8	March 1979 Well screen
20.3	March 1979 Well screen
24.8	June 1978 Well screen
32.0	March 1979 Well screen
45.0	June 1978 Fiberglass wrap

HYDR[illegible]EOLOGIC LOG

BOREHOLE No. 1814-E-1
Page 2 of 2

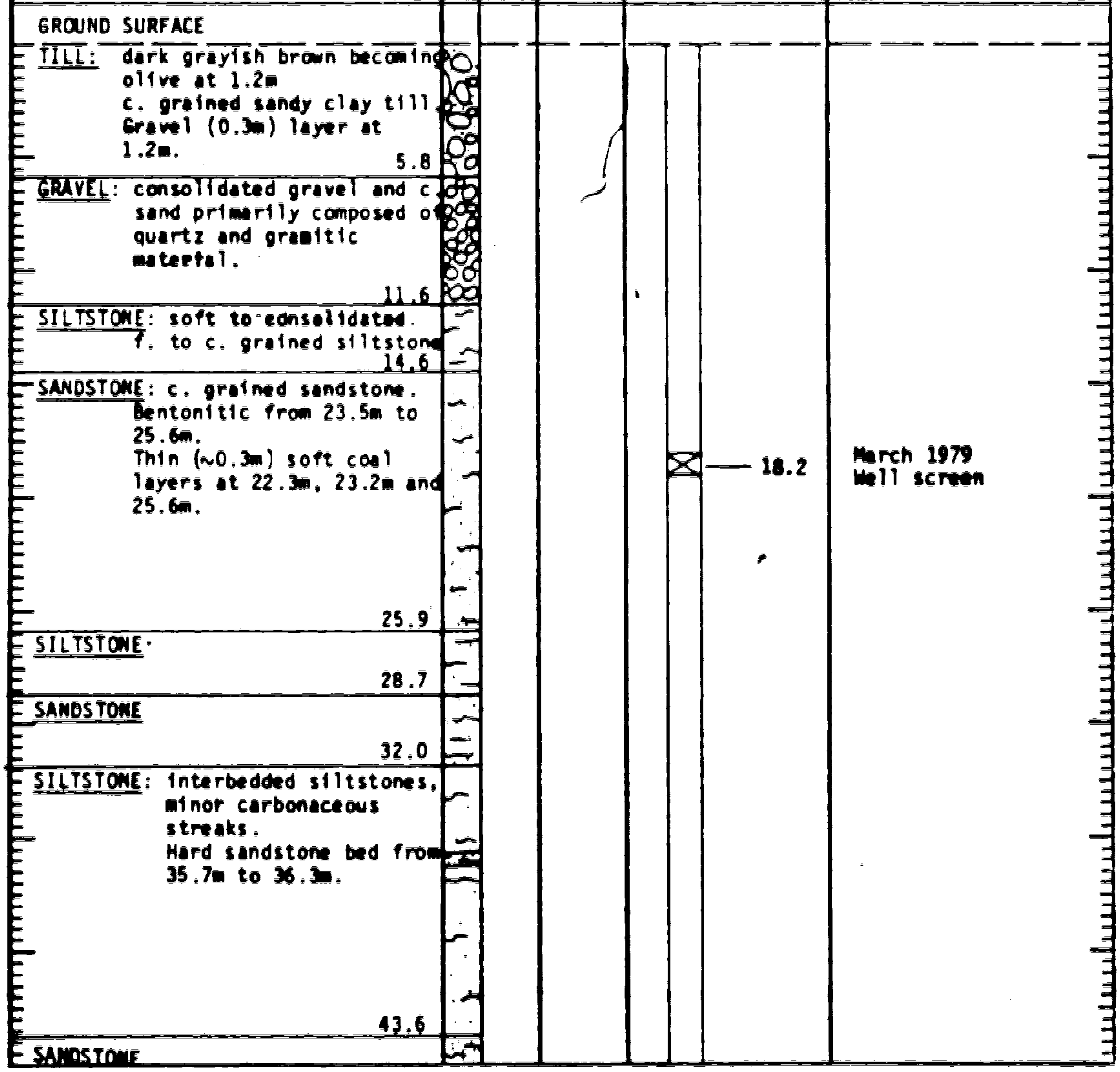
Project TABER IRRIGATION RETURN FLOW
Reference elevation 773.8
Type of drilling Rotary
Contractor Alta. Environment
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig FA 1000
Logged by M.C., et al
Drill fluid Water
Checked by
Date drilled July 1, 1978
Purpose of hole Stratigraphic control and ground water observation.

Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments
45					
SILTSTONE: continued Dark br. siltstone with thin dark br. shale beds at 44.5m, 46.5m and 54.6m. Thin coal layer at 54.6m. 54.6					
SANDSTONE: soft sandstone with 0.3m thick shale layer at 55.5m. 60.0					
SILTSTONE 61.0					
Borehole terminated at 61.0m in siltstone.					

HYDROGEOLOGIC LOG

BOREHOLE No. 1815-E-1
Page 1 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 763.2
Type of drilling Rotary
Contractor Alta. Environment.
Rig FA 1000
Logged by M.C., et al
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Drill fluid WATER
Checked by
Date drilled July 2, 1978
Purpose of hole Stratigraphic control and ground water observation.



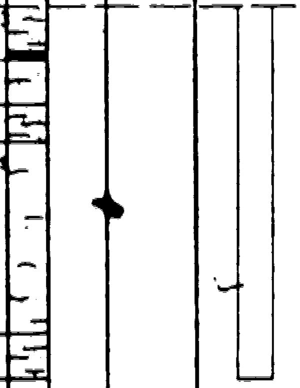
Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
GROUND SURFACE					
TILL: dark grayish brown becoming olive at 1.2m c. grained sandy clay till. Gravel (0.3m) layer at 1.2m. 5.8					
GRAVEL: consolidated gravel and c. sand primarily composed of quartz and granitic material. 11.6					
SILTSTONE: soft to consolidated. f. to c. grained siltstone 14.6					
SANDSTONE: c. grained sandstone. Bentonitic from 23.5m to 25.6m. Thin (~0.3m) soft coal layers at 22.3m, 23.2m and 25.6m. 25.9				18.2	March 1979 Well screen
SILTSTONE 28.7					
SANDSTONE 32.0					
SILTSTONE: interbedded siltstones, minor carbonaceous streaks. Hard sandstone bed from 35.7m to 36.3m. 43.6					
SANDSTONE					

HYDROGEOLOGIC LOG

BOREHOLE No. 1815-E-1
Page 2 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 763.2
Type of drilling Rotary
Contractor Alta. Environment
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig FA 1000
Logged by M.C., et al
Drill fluid Water
Checked by
Date drilled July 2, 1978
Purpose of hole Stratigraphic control and ground water observation.

Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments
45					
SANDSTONE: continued - Thin (<0.3 m) shale at 43.6m, 46.9m 47.4					
SILTSTONE 49.1					
SANDSTONE 50.9					
SILTSTONE: interbedded siltstones Hard shale layer from 58.8m to 59.4m. 59.4					
SANDSTONE 61.0					
Borehole terminated at 61.0m in sandstone.					



HYDROGEOLOGIC LOG

BOREHOLE No. 1816-E-1
Page 1 of 2

Project: TABER IRRIGATION RETURN FLOW
Reference elevation: 775.0
Type of drilling: Rotary
Contractor: Alta. Environment
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig: FA 1000
Logged by: M.C., et al
Drill fluid: Water
Checked by:
Date drilled: July 2, 1978
Purpose of hole: Stratigraphic control and ground water observation.

Lithology	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
GROUND SURFACE				
<u>TILL</u>: olive becoming dark gray sandy clay till at 9.8m. Thin coal layers at 10.7m and 25.0m. Thin gravel layers at 12.8m, 17.7m, 21.6m and 29.9m.			4.5	June 1980 Well screen
			6.7	June 1980 Well screen
			7.6	April 1979 Fiberglass wrap
			13.2	May 1979 Fiberglass wrap
			28.6	June 1978 Fiberglass wrap
30.5				
<u>SAND AND GRAVEL</u> 31.7				
<u>TILL</u>: dark gray sandy clay till, dense. Thin coal layer at 36.9m. Thin gravel layer at 35.4m.			31.9	June 1980 Well screen
40.8				
<u>SAND AND GRAVEL</u>: c. sand and gravel o occasional coal chip.			41.6	June 1978 Fiberglass wrap

HYDROGEOLOGIC LOG

BOREHOLE No. 1816-E-1
Page 2 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 775.0
Type of drilling Rotary
Contractor Alta. Environment.
Rig FA 1000
Logged by M.C., et al
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Drill fluid Water
Checked by
Date drilled July 2, 1978
Purpose of hole Stratigraphic control and ground water observation.

Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments
45					
SAND AND GRAVEL: continued 46.3					
SILTSTONE: soft siltstone 48.2					
SANDSTONE: bentonitic sandstone 50.0					
SILTSTONE: interbedded siltstones, brown carbonaceous streaks throughout. Thin coal seam at 54.3m. 56.4					
SANDSTONE: f. grained bentonitic sandstone. 59.7					
SILTSTONE: 61.0					
Borehole terminated at 61.0m in siltstone.					

HYDROGEOLOGIC LOG

BOREHOLE No. 1817-E-1
Page 1 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 773.5
Type of drilling Rotary
Contractor Alta. Environment
Rig FA 1000
Logged by M.C., et al
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Drill fluid Water
Checked by
Date drilled July 3, 1978
Purpose of hole Stratigraphic control and ground water observation

Lithology	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
GROUND SURFACE				
TILL: olive becoming grayish brn sandy clay till at 0.3m. Gravel layer from 4.0m to 4.3m. 5.8			5.7	June 1980 Well screen
~~SAND: sand with thin (0.2m) gravel~~ 7.0			7.0	April 1979 Fiberglass wrap
TILL: olive sandy clay till. Thin gravel layers at 10.4m 14.0m and 19.5m. 20.4			7.4 16.4	June 1980 Well screen March 1979 Well screen
TILL: olive gravelly clay till 22.8			21.8	June 1980 Well screen
TILL: olive sandy clay till 25.0				
TILL: olive gravelly clay till 26.8			26.7	March 1979 Well screen
GRAVEL: 29.0				
TILL: dark gray sandy clay till. 29.9				
TILL: dark gray silty clay till 31.7				
TILL: dark gray sandy clay till. Thin gravel layer at 34.7m. 37.2			32.5 36.8	March 1979 Well screen June 1978 Fiberglass wrap
GRAVEL: gravel and occasional coarse sand. 40.6				
SILTSTONE: lt. brn. to blk. siltstone.			41.8	June 1978 Fiberglass wrap

HYDROGEOLOGIC LOG

BOREHOLE No. 1817-E-1
Page 2 of 2

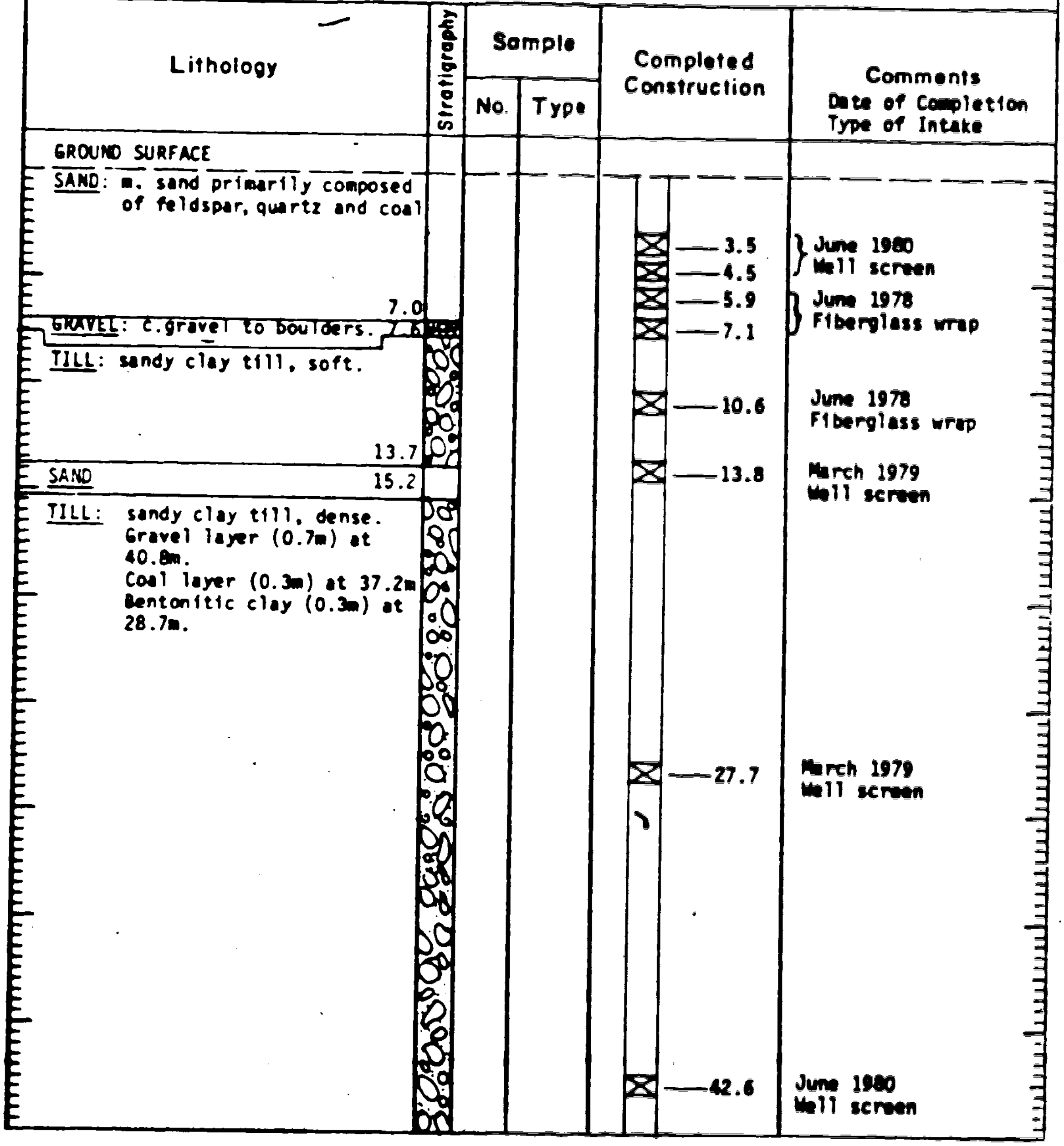
Project TABER IRRIGATION RETURN FLOW
Reference elevation 773.5
Type of drilling Rotary
Contractor Alta. Environment
Rig FA 1000
Logged by M.C., et al
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Drill fluid Water
Checked by
Date drilled July 3, 1978
Purpose of hole Stratigraphic control and ground water observation.

Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments
45					
SILTSTONE: continued Thin coal seams at 40.8m, 41.8m and 44.5m 48.5					
SANDSTONE: bentonitic sandstone. 52.4					
SILTSTONE: interbedded siltstones containing coal from 52.4m to 53.6m. Thin (0.6m) shale bed at 60.3m. 61.0					
Borehole terminated at 61.0m in siltstone.					

HYDROGEOLOGIC LOG

BOREHOLE No. 1818-E-1
Page 1 of 2

Project TABER IRRIGATION RETURN FLOW
Reference elevation 774.3
Type of drilling Rotary
Contractor Alta. Environment
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig FA 1000
Logged by DSW, et al
Drill fluid Water
Checked by
Date drilled July 4, 1978
Purpose of hole Stratigraphic control and ground water observation.

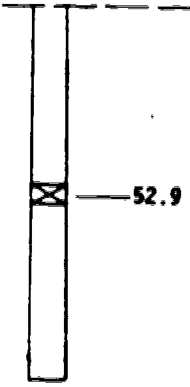


Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
GROUND SURFACE					
SAND: m. sand primarily composed of feldspar, quartz and coal — 7.0				3.5 4.5	June 1980 Well screen
GRAVEL: c.gravel to boulders. — 7.6				5.9 7.1	June 1978 Fiberglass wrap
TILL: sandy clay till, soft. — 13.7				10.6	June 1978 Fiberglass wrap
SAND — 15.2				13.8	March 1979 Well screen
TILL: sandy clay till, dense. Gravel layer (0.7m) at 40.8m. Coal layer (0.3m) at 37.2m Bentonitic clay (0.3m) at 28.7m.				27.7	March 1979 Well screen
				42.6	June 1980 Well screen

HYDROGEOLOGIC LOG

BOREHOLE No. 1818-E-1
Page 2 of 2

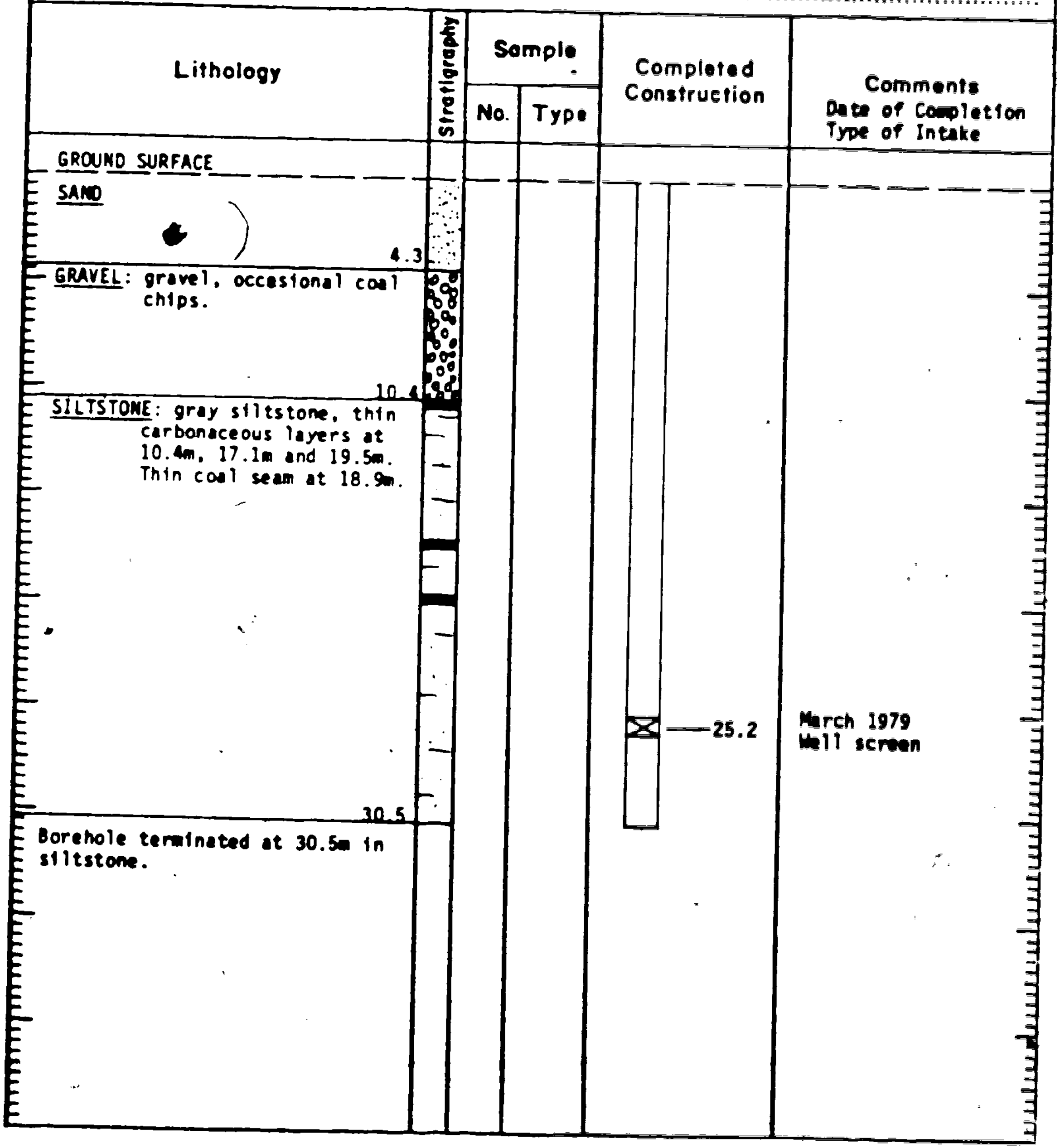
Project TABER IRRIGATION RETURN FLOW
Reference elevation 774.3
Type of drilling Rotary
Contractor Alta. Environment.
Rig FA 1000
Logged by DSM, et al
Elevation type: Serveyed ☒ Altimeter ☐ From map ☐
Drill fluid Water
Checked by
Date drilled July 4, 1978
Purpose of hole Stratigraphic control and ground water observation.

Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
45					
TILL: continued 49.7					
GRAVEL: fine to medium gravel, occasional coal chips. 53.3				—52.9	
SANDSTONE: soft to hard sandstone often clayey and bentonitic. 61.0					March 1979 Well screen
Borehole terminated at 61.0m in sandstone.					

HYDROGEOLOGIC LOG

BOREHOLE No. 1819-E-1
Page 1 of 1

Project TABER IRRIGATION RETURN FLOW
Reference elevation 737.4
Type of drilling Rotary
Contractor Alta. Environment
Rig FA 1000
Logged by D.W., et al
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Drill fluid Water
Checked by
Date drilled July 5, 1978
Purpose of hole Stratigraphic control and ground water observation.



SAND

GRAVEL: gravel, occasional coal chips.

SILTSTONE: gray siltstone, thin carbonaceous layers at 10.4m, 17.1m and 19.5m. Thin coal seam at 18.9m.

Borehole terminated at 30.5m in siltstone.

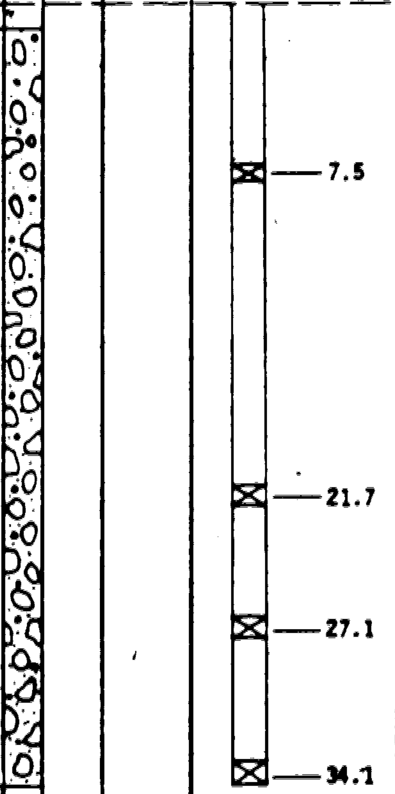
March 1979
Well screen

HYDROGEOLOGIC LOG

BOREHOLE No. 1820 - E
Page 1 of 1

Project TABER IRRIGATION RETURN FLOW
Reference elevation 773.4
Type of drilling Rotary
Contractor All-Kind
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig Failing 1500
Logged by G.B.
Drill fluid Water
Checked by
Date drilled June 1980
Purpose of hole Stratigraphic correlation and piezometer installation

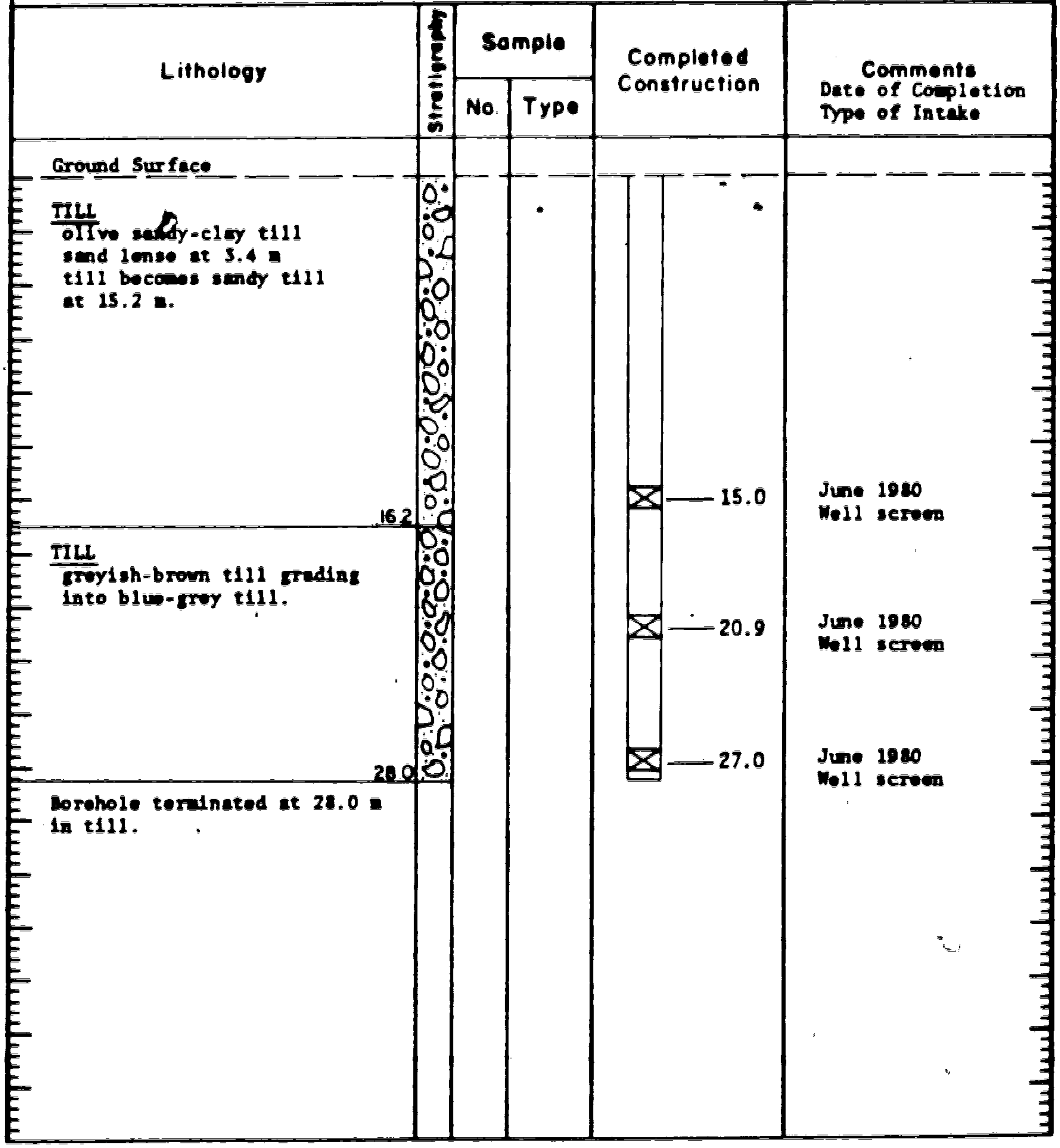
Lithology	Stratigraphy	Sample No.	Sample Type	Completed Construction	Comments Date of Completion Type of Intake
Ground Surface					
Sand and silt 1.0					
TILL olive becoming greyish-brown sandy clay till at 12.7 m. gravel layer at 17.4 to 17.9 m and 21.3 to 21.9 m. sand and gravel layer at 30.5 to 31.4 m.				7.5	June 1980 Well screen
				21.7	June 1980 Well screen
				27.1	June 1980 Well screen
34.1				34.1	June 1980 Well screen
Borehole terminated at 34.1 m in till.					



HYDROGEOLOGIC LOG

BOREHOLE No. 1821 - E
Page 1 of 1

Project TABER IRRIGATION RETURN FLOW
Reference elevation 774.2
Type of drilling Rotary
Contractor All-Kind
Elevation type: Surveyed ☒ Altimeter ☐ From map ☐
Rig Failing 1500
Logged by G.R
Drill fluid Water
Checked by
Date drilled June 1980
Purpose of hole Stratigraphic correlation and piezometer installation



TILL
olive sandy-clay till
sand lense at 3.4 m
till becomes sandy till
at 15.2 m.

TILL
greyish-brown till grading
into blue-grey till.

Borehole terminated at 28.0 m
in till.

June 1980
Well screen

June 1980
Well screen

June 1980
Well screen