H-40

Schedule B - F

Halton Region Landfill Technical Report
Site F - Burlington

The Regional Municipality of Halton

Hydrogeologic Report
Volume 1 of 3

September 1986

Trow Hydrology Consultants Ltd.

H-40

Trow Ltd

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August 28, 1986

The Regional Municipality of Halton,
1151 Bronte Road,
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Oakville, Ontario
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Attention: Ms. Susan Rogers,
Landfill Approvals Coordinator

Dear Sirs:

Region of Halton Waste Management Study
EPA Level of Detail
Site "F" Burlington

We are pleased to submit the final Hydrogeologic and Design and Operations reports for Site "F" - Burlington Sanitary Landfill.

The report is presented in three (3) volumes, namely:

Text -	Volume 1 of 3 Hydrogeologic Report for Site F - Burlington Halton Region Landfill Technical Study
Appendices -	Volume 2 of 3 Hydrogeologic Report for Site F - Burlington Halton Region Landfill Technical Study
Text -	Volume 3 of 3 Design and Operations Report for Site F Halton Region Landfill Technical Study

These reports comprise a portion of the documentation required by the Region of Halton to obtain a Certificate of Approval to operate a sanitary landfill on the subject site. The two reports are inter-dependent and should be read together. The reports cover work done by Trow Hydrology Consultants under an agreement between the Region and Trow Hydrology Consultants dated August 13, 1985 and completes the work described by that agreement.

The reports have been revised following review by Region of Halton staff and its consultants so we trust that the final product meets with your approval.

Yours very truly,
HYDROLOGY CONSULTANTS

J.P. Nunan, P. Eng.,
Principal Hydrogeologist

JJO/al

Enc.

Trow

SCHEDULE B-F

TEXT - VOLUME 1 OF 3
HYDROGEOLOGIC REPORT FOR
SITE F - BURLINGTON
HALTON REGION LANDFILL
TECHNICAL REPORT

Prepared for
THE REGIONAL MUNICIPALITY OF HALTON

HYDROLOGY CONSULTANTS
DIVISION OF TROW LTD

September, 1986

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TABLE OF CONTENTS

Trow

Text

TABLE OF CONTENTS/Continued

Trow

Text

Trow

Text

LIST OF FIGURES CONTAINED IN TEXT

Trow

LIST OF TABLES CONTAINED IN TEXT

Trow

LIST OF DRAWINGS

Trow

Drawing No.	Title
1	Location Map
2	Location of Exploratory Boreholes and Water Supply Wells
3	Typical Drilling Site Well Installation
4	Natural Surface Water Drainage and Location of Surface Water Monitoring Stations
5	Location of Drilling Sites and Observation Wells
6	Regional Stratigraphic Cross-Section A-A
7	Regional Stratigraphic Cross-Section B-B
8	Local Stratigraphic Cross-Section A-a
9	Local Stratigraphic Cross-Section C-C and D-D
10	Local Stratigraphic Cross-Section E-E and F-F
11	Bedrock Topography
12	Surficial Geology
13	Regional Bedrock Potentiometric Contour Map
14	Regional Watertable Contour Map
15	Local Bedrock Potentiometric Contour Map (April, 1986)
16	Local Watertable Contour Map (April, 1986)
17	Piper Diagram
18	Water Chemistry - Anion Profile
19	Water Chemistry - Cations Profile
20	Water Chemistry - Tritium Profile
21	Chloride Concentration Contour Map
22	Monitoring System

LIST OF APPENDICIES

Trow

Appendix 1	Summaries of Reports Prepared by Previous Investigators i) Hydrology Consultants ii) University of Waterloo iii) Morrison Beatty iv) Gartner Lee v) M.M. Dillon vi) Acres Consulting vii) Conestoga Rovers
Appendix 2	Borehole Logs i) Hydrology Consultants ii) Morrison Beatty iii) Gartner Lee
Appendix 3	Soil Analysis i) Particle Size Distribution ii) Cation Exchange Capacity iii) Total Organic Carbon iv) Chloride Content
Appendix 4	Monitoring Well Details i) Hydrology Consultants ii) Morrison Beatty iii) Gartner Lee
Appendix 5	Water Elevation Data for April 1986
Appendix 6	Meteorological Data
Appendix 7	Stream Flow Data
Appendix 8	Groundwater Hydrographs
Appendix 9	Water Well Inventory, Walk and Talk Survey
Appendix 10	Pumping Well Test Results
Appendix 11	Packer Test Results and Hydraulic Conductivity Results - Hvorslev
Appendix 12	Water Chemistry i) Surface Water ii) Groundwater iii) Enriched Tritium Data iv) Priority Pollutants
Appendix 13	Log Koc Values of Selected Organic Chemicals
Appendix 14	Retardation Factors
Appendix 15	Contaminant Migration Simulations

Trow

i) EXECUTIVE SUMMARY

In Stage 2C of the Environmental Assessment studies undertaken to locate a suitable landfill site in Halton, preliminary studies were undertaken on six potential landfill sites. As a result of considering all appropriate environmental factors, two sites, Site D in Milton and Site F in Burlington, were selected for detailed environmental evaluations. This report documents the results of detailed hydrogeologic - hydrologic studies at Site F in Burlington.

were they applied

Extensive hydrogeologic and hydrochemical studies on Site F show that with the proper engineering works, the lands proposed for landfill are hydrogeologically suitable for waste disposal.

are they superior to other sites studied?

The bedrock of Site F consists of Upper Ordovician red shales of the Queenston Formation. Four zones were identified within the shale bedrock. An uppermost Weathered or Fractured Shale is underlain by a Weakly Fractured Massive Shale Unit (calcite cemented). A Highly Fractured Shale Unit is found at the lower contact of the Weakly Fractured Shale Unit. This unit is in turn underlain by a Moderately Fractured Shale Unit which has undergone less calcite cementation.

this all seems highly permeable

weathered
weakly frac.
highly frac.
mod frac.

nice sandwich

The overburden on site is composed entirely of the Halton Till, which is silt to clayey in texture. The overburden thins from more than 18 m in the north end of the site to less than 1 m, ~~or is absent altogether,~~ towards the central portion of the site.

Groundwater movement in the upper shale bedrock is generally southeasterly towards Hamilton Harbour. There is an indication of a deep groundwater flow system at depths greater than 30 m that discharges to Hamilton Harbour. Data show "strong" downward groundwater gradients on Site F and generally lateral to weak upward gradients between drilling site HC9-85 (just south of Hwy 403) and Hamilton Harbour.

A computer simulation of the groundwater flow systems in the area indicates that leachate would migrate laterally through the Weathered Shale Unit to discharge on-site and through the underlying Highly Weathered Shale Unit to discharge off-site.

Leachate migrating laterally through the Weathered Shale Unit discharges locally and will be collected by the leachate underdrain system and treated. The leachate collection system will have a good capture efficiency and about 95 percent of the leachate produced at the site would be collected by the system. The underdrain system will also prevent toe discharges of leachate and will protect surface water from impairment.

The leachate underdrain system will not be able to overcome the vertical recharge gradients at the site and a small quantity of leachate (5 percent of the total or approximately 4.2 gpm) will exfiltrate the base of the landfill into the underlying shale units. The leachate will be attenuated by the Weakly Fractured Massive Shale Unit which has a low hydraulic conductivity and good attenuation characteristics. When the attenuated leachate enters the Highly Fractured Shale Unit it will move off-site in a southerly direction to ultimately discharge in the vicinity of the CNR tracks. In the discharge area, any remaining leachate contaminants will be further treated by attenuation as they pass through about 8 m of clayey glacial till and lacustrine sediments.

Leachate movement into the groundwater flow system is not expected to have a large impact on the Highly Fractured Shale Unit because it contains water that has naturally high concentrations of inorganic constituents such as chloride. Chloride concentrations in excess of 10,000 mg/L have been measured in this unit. Such concentrations of chloride make the water unsuitable for human consumption.

The attenuating capacity of the hydrogeologic system at Site F can be judged by observation of the two adjacent landfill sites where chemical data show no significant contaminant concentrations in groundwater beyond the limit

of the landfills. Some surface water contamination has been observed where environmental controls are absent or inadequate.

Long-term monitoring and maintenance will be required to verify the integrity of the proposed landfill design and the attenuating capacity of the hydrogeologic system.

In the unlikely event that unexpected contaminants were to escape the primary control system and the attenuating environment, site conditions are favourable for the installation of engineered systems such as perimeter tiles and purge wells to collect all water borne contaminants with potential to discharge for treatment.

The high hydraulic conductivity and lateral continuity of the Highly Fractured Shale Unit makes it feasible to install an effective purge well system that would prevent leachate from coming to surface between Site F and Hamilton Harbour.

ii) SUMMARY AND CONCLUSIONS

Hydrogeologic-Hydrologic Setting

Extensive drilling, testing and hydrochemical analysis programs were undertaken at Site F to assess the hydrogeologic suitability of the site for waste disposal. The hydrogeology of Site F has been determined and with proper controls, conditions are suitable for establishing a municipal landfill.

The dominant physiographic feature found at Site F is the deeply eroded southern edge of the Halton Till Plain. Deep gullying has modified the till plain into fluted caps overlying an incised bedrock.

Site F located about 2.5 km north of Hamilton Harbour, is situated almost entirely on the west portion of the Indian Creek basin. The headwaters of Indian Creek originate on the Niagara Escarpment. Below the escarpment, the creeks meander through a predominantly residential area and ultimately drain into Hamilton Harbour.

Bedrock in the study area consists of a series of ancient sedimentary rock formations which dip gently in a southerly to a southeasterly direction. The uppermost bedrock is Ordovician shale of the Queenston Formation.

Core logs indicate that the shale possesses a complex fracture system near the surface and at depth. The near surface fractures are the result of processes of weathering. The highly fractured zones at depth are thought to indicate planes of movement, possibly resulting from post-glacial deformation due to unloading.

Four zones within the Shale Bedrock were identified. From the shallowest to the deepest they were;

1) An uppermost weathered or fractured shale;

2) A massive weakly fractured calcite cemented shale;

3) A locally occurring highly fractured permeable shale zone; and

4) A moderately fractured shale which has undergone less calcite cementation.

The overburden on-site is composed entirely of the Halton Till. Its texture ranges from silt to clayey silt. The dominant clay mineral is illite. Overburden distribution on-site is variable due to the deep gullying action of youthful streams. The erosion is extensive enough to remove the overburden and incise the shale bedrock in stream valleys. The northern portion of the site has a more

continuous till covering. Towards Highway 403, the overburden deposits become discontinuous. South of Site F, the till is overlain by lacustrine sediments (mostly sand).

Elevated chloride and conductivity concentrations measured in the Indian Creek tributary, at the North Service Road, south of the Bayview Landfill site, indicate the presence of diluted leachate. Elevated contaminant indictors suggest that the leachate from the Bayview Landfill is degrading surface water in Indian Creek southward to at least the CN Railway.

Municipal services supply domestic water needs to residents downgradient of Site F. While it is recognized that five residents report that they still use shallow wells, existing piped municipal water is available throughout the area.

Five major hydrostratigraphic units characterize the hydrogeology of Site F. The units and their hydraulic conductivities are:

1) Overburden

   a) Till Unit at north end of site ($k = 1 \times 10^{-7}$ cm/s)

   b) Discharge Zone Till south of the site ($k = 3 \times 10^{-6}$ cm/s)

2) Fractured (Weathered) Shale Unit - highly weathered at surface ($k = 1 \times 10^{-5}$ cm/s)

3) Weakly Fractured Massive Shale Unit ($k = 1 \times 10^{-7}$ cm/s)

4) Highly Fractured Shale Unit ($k = 1 \times 10^{-2}$ cm/s)

5) Moderately Fractured Bulk Shale Unit ($k = 1 \times 10^{-6}$ cm/s)

A Highly Fractured Shale Unit has been delineated from the interpretation of core logs and pumping tests. This zone appears to be a wedge of fractured shale, underlying the Weakly Fractured Massive Unit. It is interpreted to extend from the midpoint of the landfill site to about 500 m south of Highway 403, where it contacts the overburden.

The Highly Fractured Shale Unit acts as a permeable underdrain to the upper groundwater flow system and appears to exert a major control on the groundwater flow system at Site F. Groundwater in this unit is not potable, due to naturally occurring mineralization. The chloride concentration of groundwater in this unit can exceed 10,000 mg/L. (The maximum desirable concentration for chloride in public water supplies is 250 mg/L.)

Groundwater movement in the upper 5 m of shale bedrock is generally southeasterly towards Hamilton Harbour. The horizontal gradient of the shallow bedrock flow system between the site and the harbour varies from 0.009 to 0.02.

The data indicate a deep groundwater flow system in the bedrock that probably originates above the Niagara Escarpment north of Site F. This system moves water under Site F at depths greater than 30 m and discharges into Hamilton Harbour.

On-site, the horizontal gradient of the shallow bedrock flow system ranges from 0.01 to 0.05. The direction of groundwater movement is southeasterly. The groundwater contours indicate that the groundwater in the shallow weathered shale discharges on-site into the deeply incised gullies.

The watertable in surficial materials slopes southeasterly towards Hamilton Harbour. The horizontal gradient of the shallow flow system between Site F and Hamilton Harbour varied from 0.009 to 0.02. It is interpreted that interstream zones between Site F and Hamilton Harbour are localized recharge areas that discharge to adjacent streams.

On-site, the depth to the watertable varied from about 0.15 m below ground surface at HC8-85 to about 2.9 m below ground surface at HC7-85.

beneath the glacial till, south of Highway 403. groundwater which infiltrates over the landfill area is indicated to discharge between this zone and Lake Ontario.

With respect to contaminant migration, the flow model indicates that groundwater discharge via the Highly Fractured Shale Unit would be limited to a small area (about 62 acres) south of Wells HC9-85 and HC14-85 (south of Hwy 403). The total discharge rate would be about 14,000 m^3/year (6 gpm).

Piper trilinear chemical plots of water chemistry generally indicate a dominant sulphate facies in the upper hydrostratigraphic units and a chloride dominant facies in the lower hydrostratigraphic units. Off-site, between HC9-85 and Hamilton Harbour chloride facies seem to predominate both near surface and at depth. Chloride concentrations exceeding 10,000 mg/L were measured. This distribution of dominant hydrochemical facies would indicate a slightly less mature flow system near surface at Site F underlain by a more mature geochemical systems, which becomes dominant both near surface and at depth towards Hamilton Harbour.

There is a dramatic deterioration of groundwater quality with increasing depth in the shale. The natural quality of the groundwater is poor and does not satisfy the Ministry of the Environment drinking water quality objectives.

The data show a strong downward gradient beneath Site F and generally lateral to weak upward gradients between drilling sites HC9-85 and HC14-85 (south of Site F) and HC13-85 near Hamilton Harbour.

Between Site F and Hamilton Harbour, the vertical gradients between the overburden and the shallow weathered shale are weakly upwards while the gradients between the shallow weathered shale and the deep shale formation are lateral.

A computer simulation of the groundwater flow system indicates that groundwater infiltration (or recharge) occurs from the north boundary of Site F, across and beyond the proposed landfill site, to the vicinity of drilling site HC14-85, south of Highway 403, roughly 500 metres beyond the southern border of the landfill. Much of the recharge is indicated to occur over the area of the landfill itself. The average infiltration rate is calculated to be about 14.0 mm/year, and the total volume of infiltrating water (using a site width of 500 metres) is in the order of 10,000 m^3/year (4.2 Imperial gallons per minute). Most of the groundwater that infiltrates over the proposed landfill site (and some from beyond) moves horizontally through the Highly Fractured Shale Unit.

Groundwater discharges at surface into a marsh area where the Highly Fractured Shale Unit is interpreted to outcrop

Contaminant Migration

Advection will be the principal mechanism for the transport of contaminants in the groundwater at Site F. Contaminant transport by advection is controlled by the linear groundwater velocity.

Contaminant pathway analysis and interpretation indicates that there is essentially one pathway by which leachate might migrate off-site with groundwater. The pathway consists of the following components:

a) Vertical migration downward through the Weathered Shale Unit and the Weakly Fractured Massive Shale Unit into the Highly Fractured Shale Unit.

b) Lateral migration through the Highly Fractured Shale Unit.

c) Upward migration through the Discharge Zone Till Unit

Leachate migrating laterally through the Weathered Sh Unit would discharge on-site and would be collected in underdrain system and would be conducted to treatme facilities.

The dominant hydrostratigraphic unit controlling leachate migration at Site F is the Weakly Fractured Massive Shale

Unit. Tests show that this unit has relatively good attributes for waste disposal. It has good attenuation capability and a low hydraulic conductivity (no more than 1 x 10^{-7} cm/s). Farther along the flow system, the Discharge Zone Till provides additional capability for the natural attenuation of leachate.

In order to evaluate potential environmental impacts, approximate hydraulic characteristics of a landfill were superimposed on the calibrated computer simulation of the natural groundwater flow system. The average infiltration into the underlying Weathered Shale Unit beneath the landfill increased to about 23 mm/year, for a net increase of 8.9 mm/year over the simulation for natural conditions. The discharge from the Highly Fractured Shale Unit south of Highway 403 was about 17,000 m^3/year, for a net increase of 3,000 m^3/year (1.3 gpm) above the simulated natural discharge in that area. In addition, the model indicated that there would be a leachate discharge in the order of 15,000 m^3/year (6.3 gpm) at the downgradient toe of the landfill.

The first arrival of conservative contaminants (C/Co = 0.1) at the property boundary is predicted to occur in about 30 years. Based on the estimated decline in the chloride concentration in the leachate, the maximum increase in chloride at the property boundary in the Highly Fractured

Shale Unit predicted by the model would be about 500 mg/L. This is not a severe impact because groundwater in the Highly Fractured Shale Unit has naturally high concentrations of inorganic constituents such as chloride (i.e., in the order of 10,000 mg/L) which make the groundwater unsuitable for drinking water supply. Consequently, contamination of the groundwater in itself does not constitute an unacceptable environmental impact with respect to Public Health and Safety.

Further along the flow path, the first arrival of conservative contaminants including weakly-absorbed organic compounds (C/Co=0.1 or chloride = ~~150 mg/L~~) at the Discharge Zone is predicted to occur in 60 to 100 years. The maximum increase in chloride at the Discharge Zone predicted by the model would be about 350 mg/L. This is about an 80 percent reduction in peak leachate concentration as a result of natural attenuation. This is not a severe impact since groundwater in the Highly Fractured Shale Unit is unsuitable for human consumption.

It is also considered unlikely that any other significant chemical impacts will occur for the following reasons.

1) The groundwater discharge rate at the Discharge Zone Till is calculated to be relatively low and therefore contaminant loading to the surface will also be low.

2) The contaminant model underestimates natural attenuation. Matrix diffusion and adsorption in the Weathered Shale and Highly Fractured Shale Unit are not considered by the model.

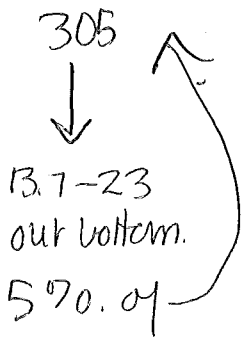
3) The hydrogeologic setting would perform well with respect to contaminant attenuation. This is evident from historic chemical data which shows no significant groundwater contamination as a result of landfilling at the two adjacent sites.

Should a problem arise, the high hydraulic conductivity and lateral continuity of the Highly Fractured Shale Unit will allow implementation of an effective purge well system to prevent leachate from coming to surface between Site F and Hamilton Harbour.

Landfill Design Considerations

A leachate underdrain system will be installed to collect the leachate produced in the landfill and thereby prevent lateral breakout at the toe. The collection system is described in the Design and Engineering Report (Volume 3).

With the disturbance of natural soils and changes to surface grades brought about by landfilling, it is anticipated that the annual infiltration of precipitation through the final landfill cover could be as much as 305 mm per annum. Most


305
13.7-23
our bottom.
570.04

of the leachate will be collected by the underdrains and transferred to an appropriate treatment facility off-site. The underdrains will not significantly affect the prevailing recharge gradients at the site and consequently a small portion of the leachate produced within the landfill will move downward through the Weakly Fractured Massive Shale Unit into the Highly Fractured Shale Unit beneath it. The quantity of leachate predicted by the model to move in this manner will be 13.7 mm to 23 mm/year. The resultant capture efficiency of the underdrain system would therefore be 95 percent.

It is estimated that about 5 percent of the leachate produced at the site would exfiltrate into the subsurface. In view of the fact that this leachate would migrate into a water-bearing zone unfit for human consumption, that a natural clay liner would only reduce the exfiltration by about 2.5 percent and that a purge well system could collect all the leachate with potential to surface at the Discharge Zone Till, if necessary, it is concluded that a liner would not provide additional environmental integrity to the site.

Contingencies

Site F would be equipped with a primary leachate collection system to protect the aquatic environment (groundwater and surface water). In addition, contingencies are available to

serve as back-ups to the primary system, namely, natural attenuation, purge wells and perimeter drains. Monitoring would determine the effectiveness of the primary collector system (underdrains) and the primary back-up system (natural attenuation). In the event that these systems prove to be less effective than expected, a secondary back-up system comprising purge wells and perimeter drains could be installed.

Purge Wells (Groundwater)

The feasibility and effectiveness of leachate control and collection by means of a purge well system has been established by the PW1 pumping test results. Based on the hydraulic coefficients determined during the pumping test, it is calculated that 3 or 4 purge wells would control contaminant migration in the Highly Fractured Shale Unit downgradient from the proposed landfill. The wells would be installed in the landfill buffer area along the North Service Road to a bottom elevation of about 118 m.a.s.l. The amount of pumpage required from the wells would be in the order of 15,000 m^3/year (7.3 gpm). The system could intercept and collect all the groundwater in the Highly Fractured Shale Unit if needed to prevent any contaminant discharge between Site F and Hamilton Harbour.

Perimeter Leachate Tile Collector System (Surface Water)

The computer simulation of a landfill superimposed on the groundwater flow system identifies the need for the primary leachate collection system. Without it, leachate seeps would develop at the toe of the landfill and move into surface water courses. The primary underdrain system will collect this leachate.

As a backup to the underdrain system, a perimeter tile collector could be installed along the edge of the landfill if monitoring indicated its requirement. The system would drain by gravity to the leachate pumping station for transmission to the Water Pollutant Control Plant.

Monitoring

Data collected from groundwater monitors located close to the landfill cells and from monitoring wells located within the property boundaries will be regularly assessed for any trends indicating adverse changes in groundwater quality. Should the groundwater quality demonstrate evidence of an unacceptable level of impairment, a contingency program would be implemented to restrain the migration of contaminants. Such a contingency can be installed within a year's time. The surface water sedimentation lagoons (see Volume 3) will be monitored routinely to ensure that the water quality is suitable for discharge to surface water courses.

Leachate quality within the landfill will be monitored in selected manholes.

The Highly Fractured Shale Unit transmits the bulk of the groundwater flow at Site F and it would be monitored routinely along with other zones to identify any off-site migration. Selected off-site private wells would be monitored annually.

Surface water monitoring stations would be established on the intermittent Creeks at Site F.

Gas collection is not believed to be necessary at this site because the site is remote from dwellings and it is generally surrounded by waste or significant buffers. However, gas monitors will be installed adjacent to all on-site structures and in the north and south buffer zones of the site.

An annual report will be prepared summarizing and interpreting all of the monitoring activities and results.

The proposed monitoring program will cover a 40-year time span subdivided into a 20-year active landfilling phase and a 20-year post closure phase or until the Ministry of the Environment is satisfied that the site poses no long-term or short-term environmental hazards.

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Shale Resource

A literature review, borehole observations and chemical test results indicate that the shale on this site is suitable for use in the manufacture of bricks and various tile products. It is also suitable for use in sewer pipe provided it is mixed with imported fire clay. It may be possible that with advanced technology such as new extrusion machines, better temperature control and uniform heat distribution, the fire clay content may be reduced.

The reserves of shale on this site could be as high as 20 million cubic meters if a full scale quarrying operation was carried out. Based on the present landfill scheme, there is expected to be about 0.5 million cubic meters or 1.2 million tonnes of excess shale which would be available for the shale industry. For comparison purposes, about 676 million tonnes of potential reserves are available in Halton (unpublished Ministry of Natural Resources report).

where would stockpile be located?

The shale from this site could be stockpiled and be available for use when required. The stockpiling procedure improves the workability of the shale since it is exposed to natural weathering.

destroy nat'l area

The presence of gypsum throughout the shale will probably necessitate the use of barium carbonate to control efflorescence. It may be possible to selectively mine the

deposit and avoid the gypsum, since it was not encountered in some boreholes. This can best be determined during the extraction period.

Potential customers for the shale are listed below. These companies could be contacted in this regard.

Brampton Brick	-	Brampton
Darlington Dunbrik	-	Mississauga
Canada Brick	-	Burlington,Mississauga
Halton Ceramics	-	Burlington
Hamilton Brick	-	Hamilton
Master Brick	-	Richmond Hill
National Sewer Pipe	-	Oakville
Original Facings	-	Toronto
Owen Sound Ledgerock	-	Toronto
Paris Brick	-	Paris

iii) STUDY TEAM

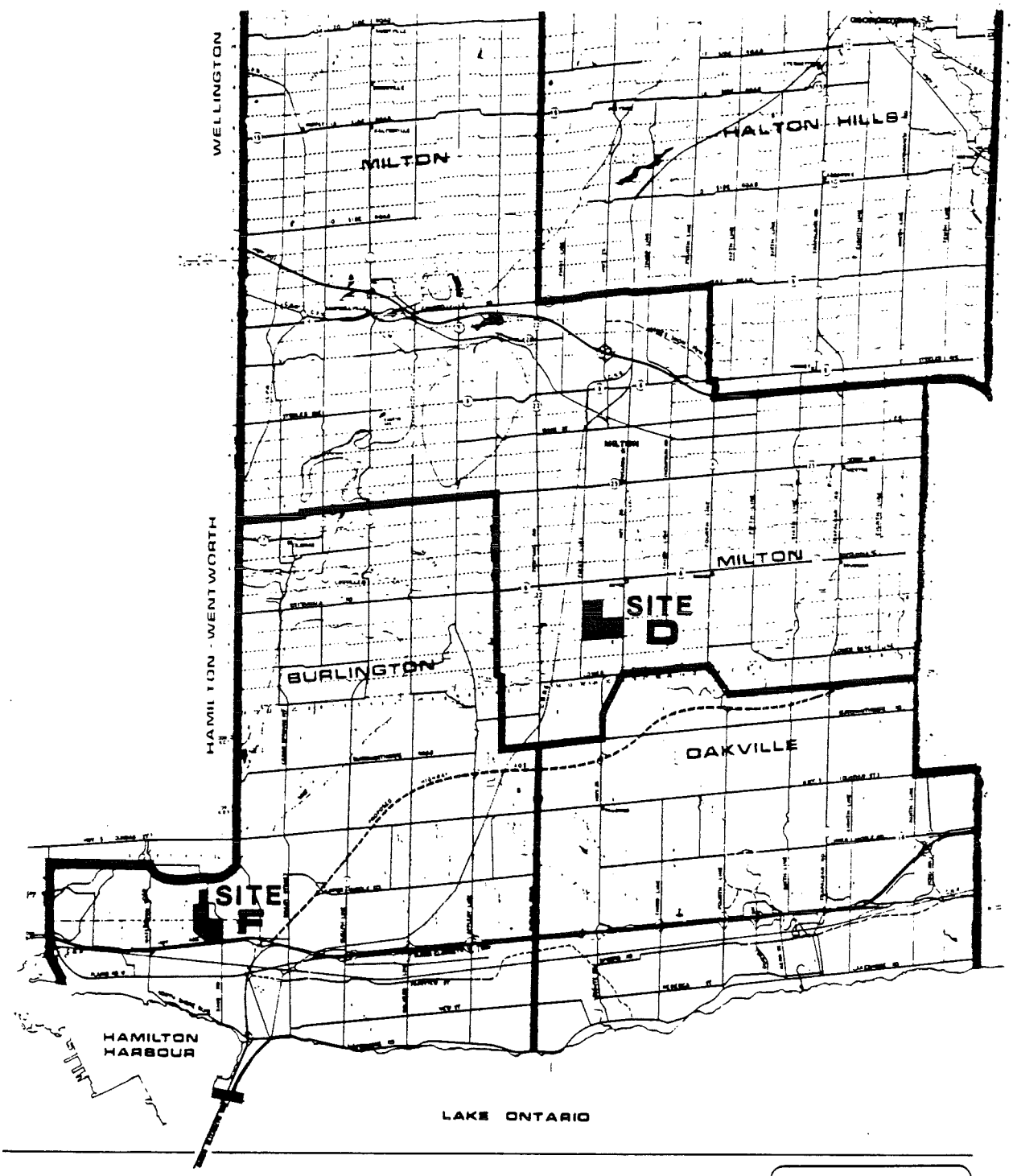
This study was carried out under the direction of Mr. J. Nunan, Principal Hydrogeologist, Hydrology Consultants.

The study team included:

R. Crysler	-	Project Manager
A. Sobanski	-	Senior Hydrogeologist
J. Beechinor	-	Hydrogeologist and Study Coordinator
J. Hiraishi	-	Senior Landfill Design Engineer
J. Ovcjak	-	Landfill Design Engineer
S. Feenstra	-	Hydrogeochemist (Zenon Environmental Inc.)
H. Lohse	-	Engineering Geologist
T. Lotimer	-	Hydrogeologic Computer Modeller
S. MacFabe	-	Groundwater Geologist
R. Carter, R. Carrington, J. Evans, S. Khan, A. Breberina, R. Zmuda, D. Odame-Osafo	-	Field Hydrogeologists, Geologists, and Hydrologic Technicians

Dr. E.A. Sudicky, of the Groundwater Research Institute, University of Waterloo, acted as advisor to the study team in regards to the application of the "FLONET" groundwater model.

iv) KEY MAP



KEY MAP

v GLOSSARY

above sea level	ASL
cation exchange capacity	CEC
centimetres	cm
centimetres per second	cm/s
chemical oxygen demand	COD
cubic feet per second	ft^3/s
cubic metres	m^3
cubic metres per square metre per day	$m^3/m^2/day$
decibels	dB
dissolved organic carbon	DOC
grams	g
hectares	ha
kilograms	kg
kilograms per cubic metre	kg/m^3
kilometres	km
litres	L,
litres per day per hectare	L/day/ha
litres per minute per hectare	L/min/ha
litres per second	L/s
metres	m
metres above mean sea level	mamsl
metres per metre	m/m
metres per year	m/yr
micrograms per litre	ug/L
micrometres	um

milliequivalent	meq
milliequivalent per 100 grams	meq/100 g
milligrams per litre	mg/L
milligrams per square metre per day	$mg/m^2/day$
millimetres	mm
million cubic metres	M m^3
million tonnes	M t
minutes	min
organic carbon content	foc
polyvinyl chloride	PVC
relative concentration with respect to input concentration	C/Co
seconds	s
square centimetres per second	cm^2/s
square metres	m^2
square metres per second	m^2/s
tonnes (metric tons)	t
Tritium units	TU
water pollution control plant	WPCP
x-ray diffraction	XRD

Slickensides	A groove or ridge on a rock surface formed by movement
Facies	Features which characterize the manner in which sediments were deposited
Vugs	A cavity in a rock formed by dissolution
Anisotropic	A substance having different physical properties when measured in different direction

vi) FORM OF REPORT

The hydrogeologic and landfill design reports for Site F - Burlington are presented in three volumes. The first volume contains the text of the hydrogeologic report, as well as figures and drawings. The second volume comprises supporting data in the form of appendices. Where an appendix is referenced, the reader is directed to the appropriate section in Volume 2. Volume 3 is the engineering report on landfill design and operations.

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1.0 INTRODUCTION

1.1 Purpose and Study Location

The overall purpose of the study was to define the hydro-geologic-hydrologic setting of Site F, with respect to its suitability to serve as a municipal waste landfill.

The site is located west of King Road and north of the North Service Road of Highway 403 in the City of Burlington, between the existing Regional Landfill site in Burlington (Burlington Landfill) and the closed Bayview Landfill site. The site includes the north half of Lots 2 and 3 north of Highway 403 in Concession I and the south half of Lot 3, Concession II of the former geographical Township of East Flamborough (see Drawing 1). Site F is approximately 96.1 ha in size.

The regional study area is generally contained in the Indian Creek and Falcon Creek basin, between Mountain Brow Road and Hamilton Harbour. This includes the two above-mentioned landfill sites. The local study area is the Site F boundary limits shown on Drawing 5. The proposed landfill area within Site F is identified in Volume 3, the Design of Operations Report.

1.2 Scope of the Study

The principal objectives of the study were:

- o to assess the hydrogeological-hydrological setting of the site;

- o to assess and predict long-term effects on water resources as a consequence of landfilling at Site F;

- o to identify measures to minimize adverse impacts to water resources; and

- o to develop hydrogeological factors and recommendations for the design and operation of a landfill at Site F.

1.3 Authorization

On July 23, 1985, the council of the Regional Municipality of Halton approved the study program necessary to establish Halton's new landfill site. Work was carried out under Purchase Order Number 68716.

1.4 Background

To date, the environmental assessment landfill site selection studies have been following a screening process to select sites suitable for landfill. This has involved progressively more detailed levels of investigations and has resulted, in Stage 2C, in the selection of preferred sites.

Subsequent to the stage 2C report, Hydrology Consultants, a Division of Trow Ltd was instructed to conduct detailed

hydrogeological investigations of Site D (Milton) and Site F (Burlington). This report documents findings with respect to Site F.

This study process is based on the requirements of the Environmental Protection Act (EPA) and is meant to resolve outstanding issues related to hydrogeology, leachate collection and treatment, landfill site design and operation, contingency measures and costs. The aim of this investigation is to provide necessary hydrogeologic information and to design an environmentally safe landfill that minimizes potential impacts on public health and safety.

1.5 Previous Work

1.5.1 General

This report makes reference to and summarizes the previous hydrogeologic studies (summarized in Appendix 1) conducted on or near Site F. These are:

(1) Hydrology Consultants, 1985 "Environmental Assessment of the Landfill Component of Halton's Solid Waste Management System Hydrogeologic-Hydrologic Assessment and Conceptual Design Report on Candidate Sites, A,B,C,D,E and F, Phase 2 - Stage 2C".

(2) Hewetson J.P., 1985, "An Investigation of the Groundwater Zone in Fractured Shale at a Landfill", University of Waterloo, M.Sc. paper.

(3) Gartner Lee and Associates Limited, 1985, "Burlington Landfill, 1985 Monitoring Report for Halton Region".

(4) M.M. Dillon Limited, 1984, "Burlington Landfill Site, Application for Continued Use," Volume 2.

(5) Conestoga-Rovers and Associates Limited and Ecoplans Limited, 1984, "Landfill Capacity and Feasibility, NSP Property and adjacent Lands, City of Burlington 1984".

(6) Acres Consulting Engineers 1982, "Solid Waste Management Strategy Study".

(7) Morrison Beatty Limited, 1981, "Phase III Hydrogeologic Investigation, National Sewer Pipe Limited Landfill Site".

(8) Morrison Beatty Limited, 1980, "Phase II Hydrogeologic Investigation, National Sewer Pipe Limited Landfill Site".

1.5.2 Hydrology Consultants, 1985

Purpose

The objective of this study was to conduct preliminary hydrogeologic - hydrologic assessments and prepare generic conceptual site plans for six candidate sites. The report was undertaken to evaluate and select sites under the Environmental Assessment Act.

Method

The general area had been studied in detail by other investigators, particularly Morrison Beatty Limited and the University of Waterloo. The data developed by these investigators were used for the site assessment. Their work was augmented by a limited drilling program undertaken by Hydrology Consultants.

Geology

The bedrock underlying Site F is red shale of the Queenston Formation which is Upper Ordovician in age. The upper few meters of the shale are extremely weathered and an extensive fracture network exists below the weathered zone.

The overburden in the upper reaches of the site consists of a dense to very dense clayey silt till. The overburden thins from more than 18 metres in north end of the site to less than 1 metre or absent altogether towards the south. Approximately 80 percent of the site lacks appreciable overburden cover.

Hydrogeology

A location map of old water wells in the vicinity of the site prepared by Gartner Lee Associates Limited indicated that there were no groundwater users downgradient of the site. Municipal services supply domestic water to residents in this area. Although some residents have reported their continuing use of the old wells, existing piped municipal water exists for this purpose.

Both the surface of the water table and the potentiometric surface generally slope south or southeastward. The horizontal gradient of this shallow flow system is 0.044. Components of the shallow flow system intermittently discharge into the north-south trending gullies at the site.

Piezometers at depths of less than 20 metres generally measure downward recharge gradients, with an average gradient of 0.3. The gradients flatten to horizontal at depths between 20 to 30 metres. Piezometers greater than 30 metres deep measure upward or lateral gradients.

Hydraulic conductivity values for the unweathered shale had a geometric mean of 9.3×10^{-6} cm/sec. From previous investigators, the linear groundwater velocity through the fracture shale was estimated to vary from 6 m/yr to 130 m/yr. Vertical velocities were estimated to be in the 0.004 to 88 m/yr range.

Previous reports indicated that there is a dramatic deterioration of groundwater quality with increasing depth in the shale. Natural quality of the surface water is poor. Because of these factors, any contaminant plumes from the existing Burlington and former Bayview landfills were difficult to identify.

Conclusions

There were some divergencies of opinion among investigators regarding groundwater movement in the shale and the extent of existing contamination from neighbouring landfills. They did, however, agree that the natural groundwater quality is poor and at depths of 15 m in the shale's background groundwater quality does not meet current Ministry of the Environment drinking water objectives.

The area downgradient from the site is serviced with a municipal water supply system.

Other reports are summarized in Volume 2, Appendix 1.

2.0 STUDY METHODS

2.1 General Approach

The study methods were intended to resolve outstanding issues related to hydrogeology. It was necessary to establish the significance and characteristics of important hydrogeologic features in order to assess the potential impact of landfilling on the environment.

Of concern, at Site F, was the potential for leachate discharge between Site F and Lake Ontario (Hamilton Harbour), as a consequence of leachate infiltration into the shale at Site F. The Ministry of the Environment stipulates that leachate infiltrating into the shale should not surface between Site F and Hamilton Harbour.

2.2 Liaison with Concerned Parties

It was the intent to develop a fieldwork program, generally agreed to by the Ministry of the Environment, by the Halton staff and their advisors, by member Municipalities and by Hydrology Consultants. To this end,

i) The detailed description of the fieldwork program, "Activity 1A, Drilling and Well Installation for the Detailed Hydrogeologic Assessment, August, 1985" was circulated;

ii) The ideas and general views of the staff of the area municipalities and their consultants were received and considered; and

iii) The Ministry of the Environment and Halton staff and their advisors were consulted to obtain their comments with respect to the Hydrogeologic work program for Site F.

The review process was scheduled to be completed during the first week of September before the fieldwork commenced. However, because of the slow response of some of the intervenors, this process took longer than expected. As a result, some suggestions were received after the fieldwork was completed.

As a result of the field program undertaken in Phase 1, and the liaison with concerned parties, additional fieldwork requirements were identified. To satisfy these requirements, an extended Phase 2 field program was initiated in February, 1986. Some of the important concerns to which response was made are shown in the "Party - Comment Matrix" in Figure 1.

Trow

FIGURE 1

PARTY - COMMENT MATRIX

Site F

PARTY

Comment	City of Burlington Conestoga - Rovers & Ass. Ltd. (CRA)	Ministry of the Environment	Ministry of Natural Resources	Responses or Actions of Hydrology Consultants
Review of shale quality detailed work program			*	MNR provided with detailed program
Utilization of shale prior to landfill development			*	Shale testing carried out for this purpose; shale resource usage was considered
Insufficient number of suitable wells for groundwater sampling	*			Sampling program expanded to include wells at HC8-85 and several off-site wells (well construction was modified accordingly)
Sequential packer test at a borehole	*			This was done on borehole HC6-85
Locate drilling site HC9-85 near MB16	*			This was done
Test pits to assess shale workability	*			6 test pits were excavated for this purpose
Separate boreholes for each well monitor; seal wells with cement grout	*			Seals were tested for competency, if not competent, wells were re-installed. Thick bentonite seals were used because cement grout can have a long-term effect on water chemistry
CRA field observer on-site	*			Attended CRA observers on-site
Chemical stability of well water	*			Repeated measurement of pH, conductivity and chloride to establish the chemical stability of well water
Precipitation patterns for study period		*		This data was collected and analyzed
Pumping test to define characteristics of permeable zones in shale	*	*		Pumping tests were conducted at 3 locations
Assimilate work carried out at Burlington and Bayview landfills		*		Work of other investigators has been assimulated into this study
Leachate infiltrating into shale should not come to surface between site and Hamilton Harbour		*		Hydrogeology between site and Hamilton Harbour was investigated; the potential for leachate outbreaks was addressed

2.3 Finalized Fieldwork Program

2.3.1 Phase 1

Fieldwork was initiated in September, 1985, when the firm of All-Terrain Drilling Limited and Master Soil Investigations Limited commenced drilling operations at the site.

Eight drilling sites were selected and several boreholes were advanced at each drilling site with hollow stem augers by means of a CME-75 drilling rig. The overburden was drilled using hollow stem augers and the bedrock was cored (air and water rotary methods). In addition, boreholes were advanced at five drilling sites between Site F and Hamilton Harbour (HC9-85 to HC13-85).

At most drilling sites, the deepest borehole was drilled 30 m below ground surface.

At each drilling site, a continuous soil sampler was used to sample the complete overburden sequence, and a continuous core sampler collected the complete bedrock sequence to 30 m. The bedrock on-site was drilled using air-rotary techniques in order to accurately analyze groundwater quality. This ensured that foreign water was not introduced into the boreholes.

In the field, hydrogeologists and geologists examined the soil samples recording information such as soil moisture,

soil colour, soil weathering and general soil texture. As well, geologists examined the rock cores recording such information as joint and fracture characteristics, fracture frequency and rock strength. Soil and rock samples were carefully preserved in foil, plastic wrap, PVC core trays and wooden core boxes. Samples were transported to Trow's Soil Laboratory (Brampton). At the laboratory, the samples were re-examined by senior staff. Any discrepancies between the two logs was resolved by further examination and final borehole logs were prepared.

Packer injection tests were conducted on permeable borehole zones encountered at drilling sites HC3-85 and HC4-85 in order to establish the permeability of the shale. The test involved the sealing of the section of borehole to be tested by inflating packers and injecting water into the bedrock formation. The volume of water moving into the formation versus time is indicative of the permeability of the zone tested in the bedrock. The deep borehole (30 m) at HC6-85 was sequentially packed off in 3.0 m intervals and tested over its entire depth.

The equation for hydraulic conductivity of the rock mass using packer testing (U.S. Bureau of Reclamation, 1965) is as follows:

$$K = \frac{Cp \cdot q}{h}$$

where Cp = Relationship between diameter of borehole and length of test section

q = Constant rate of flow into the borehole

h = h(gravity head) + h(applied pressure)

The location of the thirteen drilling sites are shown on Drawing 5 (HC1-85 to HC13-85). The borehole logs are documented in Appendix 2. At each drilling site, two or more boreholes were drilled, and subsequently were instrumented with three or four observation wells. For example, four observation wells were installed at drilling site HC8-85.

Drilling Site	Observation Well	Well Type	Well Depth (m)
HC8-85	1 (lowest)	piezometer	30.9
	2	piezometer	18.6
	3	piezometer	11.6
	4	standpipe	6.9

The wells were constructed with 32 mm diameter, threaded, flush jointed PVC pipes. PVC well screens with a slot size of 0.15 mm were used in all wells. Each drilling site consisted of: 1) a single shallow standpipe (about 5 m below ground surface) continually screened for the full length of the pipe to about 1 m below ground surface; 2) one or more intermediate depth piezometers (10-20 m); and 3) a

deep piezometer (up to 30 m below ground surface). At most drilling sites, the standpipe was installed in the overburden and piezometers 2 to 3 were installed in the shale bedrock. Drawing 3 illustrates a typical drilling site well installation.

The primary function of these wells was to allow groundwater level monitoring and gradient calculations. These wells were also used for groundwater sampling and in-situ hydraulic conductivity testing.

The piezometer well screens were set in medium-grained sand that covered the screened interval. The sand was capped with 3 m thick bentonite seals. Sand and gravel was employed to fill the remaining annulus interval volumes and a mounded surface seal of bentonite and auger cuttings was used to prevent surface water from ponding around the installations. Locked protective casings were installed around the observation wells. Well construction details are documented in Appendix 4.

see Drawing 3 diagram

An engineer's survey to determine the elevations of the boreholes and the observation wells was carried out.

Water recovery tests were conducted on all of the observation wells. The Hvorslev Method (1951) was used to estimate the hydraulic conductivity (K) of the overburden and bedrock

using the variable head formula for a well point filter at an impervious boundary. The equation for hydraulic conductivity is as follows:

$$K = \frac{d^2 \ln \frac{(2L)}{D}}{480\ L\ (t_2 - t_1)} \ln \frac{H_1}{H_2}$$

where
- D = diameter of intake (cm)
- d = diameter of piezometer (cm)
- L = intake length (cm)
- H_1 = depth to water below static water level at t_1
- H_2 = depth to water below static water level at t_2
- t_1 = time at H_1 (min)
- t_2 = time at H_2 (min)

The calculated hydraulic conductivities are summarized in Appendix 11. A typical calculation is also shown in Appendix 11.

Groundwater levels were measured in all the observation wells installed on-site twice a month for eight months. Groundwater hydrographs of the data are shown in Appendix 8. The April 1986, measurements employed in the hydrogeologic interpretations are found in Appendix 5.

Observation wells sampled for background groundwater quality (wells at drilling sites HC1-85, HC4-85, HC5-85 and HC8-85, HC9-85, HC10-85, HC11-85, HC12-85 and HC13-85) were developed and purged with a bailer. Six well casing volumes of water were removed from wells prior to the sampling. A triple-tube sampler was used to collect groundwater samples from WAT 2, the University of Waterloo background quality well north of the Bayview Landfill.

Surface water samples were collected at various locations along the reaches of Indian Creek. These locations are shown on Drawing 4. The Indian Creek Valley between the Site F and Hamilton Harbour was inspected for bedrock outcrops.

Test pits were dug at locations shown on Drawing 5 to investigate the weatherability and rippability of the shale.

2.3.2 Phase 2

The Phase 2 field investigation commenced February 1986. It involved more drilling and the installation of additional observation wells off-site at (HC14-86, HC15-86, HC16-86) in order to better define the local and regional hydrogeology. The need for more thorough understanding of the groundwater flow systems in the area south of Hwy 403 contributed to the development of the Phase 2 Work Program. Groundwater level monitoring and in-situ hydraulic conductivity tests were

conducted on all of the new observation wells. The locations of the drilling sites are shown in Drawing 5 and borehole logs in Appendix 2.

In discussions with Gartner Lee Associates Limited, it was evident that in addition to data required for this study, the well installations at HC15-86, south of the Burlington Landfill could also provide valuable water chemistry data in connection with Gartner Lee's plume delineation study of the existing Regional Landfill site in Burlington. For this reason, stainless steel wells were installed at HC15-85, in order to match the well construction material used for the plume delineation study.

In February 1986, three pumping tests were conducted on the shale bedrock. Test locations are denoted PW1, PW2, and PW3 in Drawing 5. Northern Well Drilling Limited (Newmarket) conducted the pumping tests under the supervision of Hydrology Consultants.

Two test wells were installed at each pumping test location to investigate:

- the hydraulic and water yielding characteristics of the shale;

- the lateral continuity of the fracture zones; and

- the feasibility of installing a purge well contingency system.

Pumping well PW1 and its observation well were installed to a depth of about 30 m below ground surface near the North Service Road and King Road. The bedrock in both wells were uncased over a distance of about 11 m. The bedrock was subjected to a 24-hour pumping test followed by about 22-hours of recovery. In addition to the drilled observation well, water levels in nearby wells installed by the University of Waterloo and Morrison Beatty Limited were monitored during the test. The results of the tests are discussed and reported in Section 6.0 and Appendix 10.

More permeable water-bearing zones were identified in the shale at HC3-85 as a result of the packer injection tests. A water-bearing zone was also identified by work carried out by the University of Waterloo at WAT 5. To estimate the horizontal continuity of these zones, pumping tests were carried out at these locations.

Near HC3-85, PW2 and its observation well were installed to a depth of about 16 m below ground surface. Both wells were constructed as open holes over a length of about 8 m. Water levels in wells at HC3-85 and HC4-85 were also monitored during the test. The bedrock at this location was subjected to a 24-hour pumping test followed by about 22-hours of

recovery. Results are discussed and reported in Section 6.0 and Appendix 10.

Near WAT5, PW3 and an observation well were installed to depths of about 7.5 and 10 m, respectively. Their open hole intervals were about 2.5 m (PW3) and 3 m (observation well). The bedrock at this location was subjected to a 7-hour pumping test followed by about 17-hours of recovery. Results are discussed and reported in Section 6.0 and Appendix 10.

2.4 Laboratory Analysis

2.4.1 Soil

Selected soil samples were subjected to sieve and hydrometer analyses at Trow Geotechnical Laboratories. The grain size distribution curves of the soil samples tested are shown in Appendix 3.

The percent calcite and dolomite of samples collected from HC1-85, clayey silt till, and HC4-85, shale, were determined at the University of Guelph, Department of Land Resource Science. The amount of calcite and dolomite expressed as weight percent of the whole soil was determined gasometrically using the "Chittick" apparatus (Dreimanus, 1962). Results are discussed in Section 4.2.

Clay mineral identification of the $< 2\ \mu m$ fraction was determined by x-ray diffraction (XRD) analysis. Because of the large amount of carbonates present in the samples, each was treated with a sodium acetate-acetic acid buffer (pH 5.0) prior to separation of the clay fraction. Treatments performed on the clay fraction during XRD analysis were:

- Mg^{++} saturation;
- Mg^{++} saturation, glycol solvation;
- Mg^{++} saturation, glycerine solvation (if needed);
- K^{+} saturation;
- K^{+} saturation, heating to 375°C for 1 hour;
- K^{+} saturation, heating to 550°C for 1 hour; and
- HCL treatment, K^{+} saturation

The samples were run on a Rigaku D/Max IIA XRD system. Results from the XRD analysis are listed in Figure 4, Section 4.2.2. All results are qualitative.

Two common factors used to assess the leachate retardation capacity of overburden and bedrock material are its cation exchange capacity (CEC) and its organic carbon content (foc).

The cation exchange capacity of a shale or clay overburden material is related to the type and structure of the clay particles. As indicated by the diagnostic name, cation

exchange capacity reflects the ability of a geologic material to accept and exchange cations (positively charged ions) from the groundwater solution. Cation exchange capacities are reported in units of milliequivalents per 100 g of dry sample (Freeze and Cherry, 1979). Soil samples were obtained from drilling sites HC2-85 and HC10-85. These samples were submitted to Zenon Environmental Inc. (Burlington) for cation exchange analysis. The total CEC of the shale at Site F was determined using a sodium-saturation-magnesium elution method. Results are discussed in Section 8.4 and are reported in Appendix 3. The CEC of the upper till determined at Site D is also considered representative of the CEC of the clayey silt till at Site F.

The organic carbon content of the bedrock reflects the quantity of organic matter present in the shale. The presence of organic carbon can serve to retard the migration of organic substances through the mechanism of sorption. Rock samples obtained from drilling sites HC3-85, HC4-85, HC7-85, HC8-85, HC10-85 and HC12-85 were submitted to Zenon Environmental Inc. (Burlington) in order to establish their organic carbon content. Results are discussed in Section 8.4 and are reported in Appendix 3. The foc contents of the shale were measured using a sulfuric acid - dichromate digestion method. The foc of the upper till determined at Site D is also considered representative of the clayey silt till at Site F.

The chloride content of the shale rock mass was analyzed by Zenon Environmental Inc. Rock samples were collected from HC7-85, HC8-85 and HC12-85. The analytical results are shown in Appendix 3.

To evaluate the shale with respect to its suitability for use in sewer pipe or brick manufacturing, cores were examined to confirm the shale lithology and to establish an estimate of the amount of deleterious substances such as gypsum. Selected samples of the shale from the upper 10 m were subjected to chemical analysis so that the characteristics of the shale could be compared with published data from nearby active quarries. The chemical composition of the shale is shown in Table 3.

2.4.2 Water

Eight groundwater samples were collected from wells at HC5-85 and HC8-85 and were sent to the University of Miami for the analysis of enriched Tritium. Results are reported in Appendix 12 and are discussed in Section 7.4. This information was useful in determining the depth of penetration of recent waters into the soil and rock profile and therefore, assisted in the assessment of the potential rate and mode of leachate migration.

Prior to 1952, the natural Tritium content of precipitation was estimated to be between 5 and 20 Tritium Units (TU).

Beginning in 1952, the above ground testing of thermonuclear bombs resulted in an increase in the Tritium activity in the atmosphere. Post-1952 precipitation contains a greater Tritium concentration (in 100's of TU) and so this isotope can be used to delineate pre-1952 waters from post-1952 waters.

To establish background surface water and groundwater quality, five surface water and thirty groundwater samples were tested for general chemistry parameters including pH, hardness, alkalinity, calcium, magnesium, potassium, conductivity, iron, chloride, phenolics, sulphate and dissolved organic carbon. The pH and conductivity of the waters were tested in the field, while the remaining parameters were analyzed in the laboratory by Barringer Magenta (Rexdale). Results are found in Appendix 12.

Prior to this sampling program, well water was tested for pH, conductivity, chloride and sulphate on two occasions (November 1985 and January 1986). This preliminary testing was necessary to establish the chemical stability of well water. Well water was judged to be chemically stable when sampled.

At the request of Conestoga Rovers Associates Limited, observation wells at HC9-85 were installed beside the nest of wells installed by Morrison Beatty Limited (MB-18) in

order to confirm previous water chemistry results. Trace organic constituents in groundwater measured at MB-18 had been used by the University of Waterloo to delineate the extent of landfill - derived contaminants from the Bayview Landfill.

A groundwater sample was collected from observation well No. 3 installed at the upper shale - till contact at HC9-85 (about 9.5 m below ground surface) and was analyzed for a wide range of priority pollutant organic compounds by Barringer Magenta Limited. Results are shown in Appendix 12.

Groundwater samples collected from the stainless steel wells installed at various depth in the shale at HC15-86 were also analyzed for a wide range of priority pollutant organic compounds by Zenon Environmental Inc. Results are shown in Appendix 12.

Groundwater samples collected from WAT 2 were examined for several inorganic chemical parameters (see Appendix 12) in order to document ambient groundwater quality.

2.5 Inventory of Water Use

In September, 1985, "a walk and talk" survey of water well users in the vicinity of Site F was carried out. The survey area and the locations of groundwater supply users are shown on Drawing 2. In all, 14 residents were interviewed.

Information regarding local geology and water use was obtained during the survey. The results of the interviews, along with other water use information collected from the Ministry of the Environment are compiled in Appendix 9. Local land-use activities potentially detrimental to water resources were also identified (if any).

2.6 Hydrogeologic and Geologic Relationships

Field data were interpreted to assess the potential for contaminant migration in the subsurface at Site F. This information was related to the conditions of the neighbouring hydrogeologic setting and local water supply sources. In addition, Trow Hamilton's Geotechnical files for borehole information were examined in order to verify stratigraphic interpretations between Site F and (Hamilton Harbour) Lake Ontario. The following specific activities were carried out:

a) Two regional and five local stratigraphic cross-sections of the site were prepared. These sections were used to determine or show:
   - depth to bedrock
   - composition and thickness of overburden
   - distribution of fractures and joints in the Queenston shale
   - potentiometric and water table levels
   - homogeneity and continuity of overburden deposits

- key hydrostratigraphic units

b) Potentiometric and water table levels were measured in observation wells and vertical gradients were calculated to determine areas of groundwater recharge and discharge in the vicinity of Site F.

c) Shallow standpipes were used to measure the approximate depths of the water table. The water table is defined as the surface on which the fluid pressure in the pores is at atmospheric pressure. Lines of equipotential were constructed from the elevations of the water table measured in the shallow standpipes. Shallow groundwater flow is at right angles to lines of equipotential.

d) The direction of groundwater movement in the shale was approximated from static levels in the deeper piezometers in a manner similar to that described above.

e) Computer model studies were undertaken to provide a more detailed analysis and understanding of the existing groundwater flow system at Site F. A computer model, "FLONET", developed by Frind and Matanga (1985) was used. This program solves for the hydraulic head distribution and flow line distribution within a two-

dimensional porous media domain. The model was used to simulate the existing groundwater flow system. The model was calibrated to approximate the actual field data, including measured hydraulic conductivities, interpreted hydrostratigraphy and April 1986, water table levels.

To evaluate the effect of a landfill on the groundwater flow system, a computer simulation was completed assuming the presence of a landfill at Site F.

f) The chemical results were interpreted.

g) The potential of contaminant migration through the various hydrostratigraphic units at Site F were assessed using a series of one-dimensional analytical models to simulate the effects of advection, hydrodynamic dispersion, diffusion and attenuation due to adsorption. All component models were calculated using an Apple MacIntosh micro-computer with Excel spreadsheet-graphics software.

h) Measures to control the degradation of off-site water resources were identified and are discussed herein.

i) A program was established to monitor contaminant movement in order to identify problems before water resources are adversely effected.

j) The impact of the loss of the shale extraction at Site F as a consequence of the landfill was investigated.

2.7 Sources of External Information

A review of selected literature was conducted focusing on groundwater resource, regional stream flow characteristics, water well records, overburden and bedrock geology and soil survey information. Stereo air photo coverage (1985) was used to detail surficial features.

Personal contact with several government agencies provided data on climatic, water wells and geologic interpretations of the local area.

The sources of the external data are referenced in Section 17.0.

3.0 PHYSIOGRAPHY AND TOPOGRAPHY

3.1 Physiography

Site F is generally situated in the Physiographic Region defined by Chapman and Putnam (1984) as the South Slope. In the study area, this physiographic region is characterized as a narrow belt of undulating glacial till situated above the Iroquois shorecliff. The dominant physiographic feature found at Site F is the deeply eroded southern edge of the Halton Till plain (see Drawing 12).

South of the Lake Iroquois shoreline, lacustrine sediments overlie the Halton Till Plain.

3.2 Topography

The surface relief found at Site F could be described as extensively dissected by ravines. This deep gullying has modified the till plain into fluted caps overlying an incised bedrock. Stream valleys show a "youthful" stage of development represented by straight courses, with most of erosion occurring downward.

4.0 GEOLOGY

4.1 Regional Geology

Stratigraphic cross-sections A-A and B-B (Drawings 6 and 7) show the regional relationship between the overburden and the bedrock in the vicinity of Site F.

4.1.1 Bedrock

Bedrock in the study area consist of a series of ancient sedimentary rock formations which dip gently in a southerly to a southeasterly direction. The study area is underlain by Upper Ordovician shales of the Queenston Formation. This unit is regionally considered to be undisturbed tectonically, however, local disturbances within the Queenston formation have been attributed to post glacial stress relief. The regional bedrock topography is shown on Drawing 11.

4.1.2 Overburden

Two major overburden units exist in the study area, namely the Halton Till and lacustrine sediment (see Drawing 12).

The Halton Till, a silt to clayey silt till, overlies the bedrock except where the overburden is completely removed. Its deposition is considered to be the result of the last advance of Late Wisconsinan ice to the northwest out of the Lake Ontario Basin (Karrow, 1963).

When the ice had retreated some distance to the east, Lake Iroquois was formed and prominent lake features were developed (Karrow, 1963). The deposition of shallow water lacustrine sediments during this time is represented as the second overburden unit. This unit overlies the Halton Till from the old Lake Iroquois shoreline and thickens toward Lake Ontario.

4.2 Local Geology

An interpretation of the local geology is portrayed in stratigraphic cross-sections denoted A-a, C-C, D-D, E-E and F-F (see Drawings 8 to 10).

4.2.1 Bedrock

The shale at the site is the Queenston Formation of Upper Ordovician Age. It consists of brick red thin to thick bedded sandy and argillaceous shale (Guillet, 1967). The lower contact of the formation is reported to contain occasional greenish and reddish limestone. The formation has seams or bands of green colored shale throughout. At some locations the shale has a mottled appearance where both the red and green colour occur in patches. On Site F the average proportion of green beds is about 20 percent. Chemically the green beds contain less Fe_2O_3 (Hematite) than the red beds.

According to Guillet (1967) the Queenston Formation becomes progressively less siliceous and more dolomitic from bottom to top. The content of iron is constant throughout but the lime content varies irregularly. Gypsum occurs as nodules of less than 25 mm in diameter, botryoidal (rounded) masses and tabular sheets. Cavities in the shale are attributed to the dissolution of gypsum. On Site F, both gypsum nodules and tabular sheets along joints were noted to occur in pockets, but these are estimated to comprise less than 1 percent of the rock. Much of the shale was observed to be free of gypsum.

A review of the core logs taken over the site indicates that the shales possess a complex fracture system near surface and at depth. Exposure of the bedrock on the site is a consequence of the extensive removal of overburden material due to erosional and gullying processes occurring at surface.

Table 1 identifies the clay-size fraction of the bedrock. Illite is the dominant clay mineral in the Queenston Formation at Site F.

TABLE 1

MINERAL IDENTIFICATION
OF THE CLAY-SIZE FRACTION IN BEDROCK

Borehole	Depth Interval metres	Mineralogy, 2 um Fraction
HC4-85	0-1.5	Illite - major (perhaps 2 to 3 times as much as illite) Chlorite - lesser
HC4-85	6.0-7.0	Illite - major Chlorite - lesser Interstratified Clays - present Quartz - small amount

4.2.2 Overburden

The overburden on Site F is composed entirely of the Halton Till. Its texture ranges from silt to clayey silt. Based on the analysis of a soil sample collected at HC1-85, the dominant clay mineral in the Halton Till is illite (see Appendix 3).

Overburden distribution on-site is greatly modified due to the deep gullying action of youthful perennial streams. The resultant topography shows linear ridges of Halton Till. The erosion is commonly extensive enough to remove the overburden and deeply incise the shale bedrock. The northern extent of the site possesses a more continuous till covering. Towards Highway 403, the overburden deposits become discontinuous.

what is the percentage

4.3 Discussion

Initial review of the geologic configuration for the site seems to reflect a simple system of shale bedrock overlain by a cover of clayey silt till. The complexity of the geologic system lies not with the surficial features but with the fracture characteristics at depth.

A detailed study of the core logs resulted in the delineation of discrete zones of fractured and weakly fractured shale (see Figure 2). By implying lateral connection of these features, (as shown by pumping tests, refer to Section 6.2.4) a general fracture network was interpreted. Russell and Harman (1984), noted in their borehole studies at Milton, that the Queenston Formation possesses large soft sediment deformation shears (slickensides) and that their apertures are a consequence of stress release. Russell and Harman state that it is reasonable to regard these features as significant contributors to the permeability of this unit. The observed frequency of these slickensides in the Queenston Formation appear to be controlled by the degree of diagenetic (post-depositional) cementation serving to "weld shut" these compactional shears. An observed trend (Russell and Harman) in the slickenside fractures show high frequency at the top and bottom of the unit with low frequency in the middle. This trend correlates with increased point load strength and calcium content where the slickenside frequency decreases. (See Figure 3).

FIGURE 2

LOCAL CATEGORIZATION OF THE
QUEENSTON FORMATION
BASED ON DISTRIBUTION OF FRACTURES

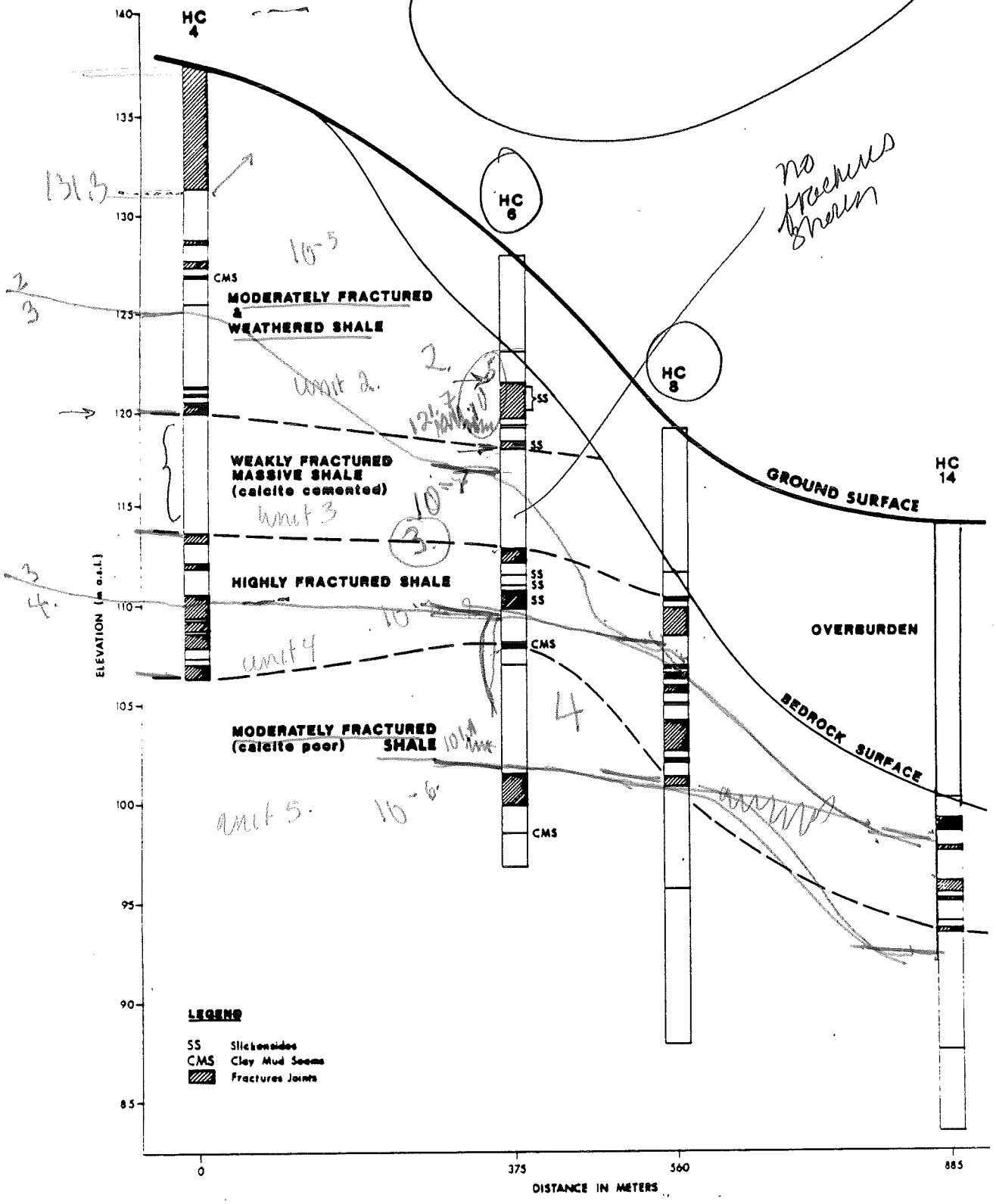
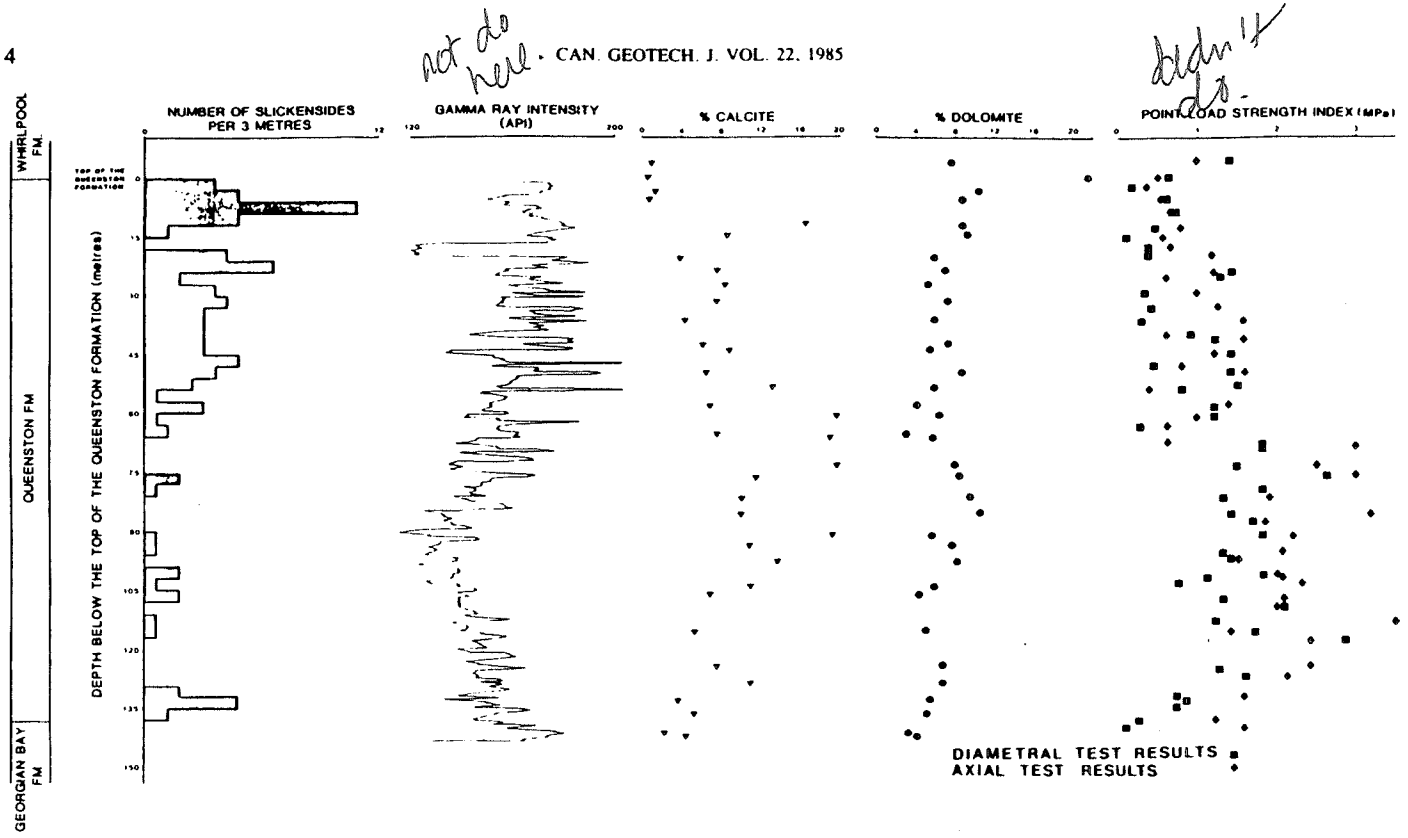


FIGURE 3

DEPTH PLOTS FOR QUEENSTON FORMATION FROM MILTON BOREHOLE OF
(a) SLICKENSIDE FREQUENCY,
(b) GAMMA RAY INTENSITY,
(c) CALCITE AND DOLOMITE CONTENTS, AND
(d) POINT LOAD INDEX FOR AXIALLY AND DIAMETRICALLY LOADED SAMPLES.

(RUSSELL AND HARMAN, 1984)



The results of the limited calcite and dolomite analyses performed at Site F, show approximately twice as much calcite and dolomite expressed in percent by weight in the massive shale as in the weathered shales (see Table 2). The trend observed by Russell and Harman appears to occur at Site F. Less competent zones in the shale (weathered shale) have relatively low carbonate content.

TABLE 2

CALCITE AND DOLOMITE ANALYSIS
USING THE CHITTICK APPARATUS

Borehole	Depth Interval metres	Sample Description	% Calcite by weight	% Dolomite by weight
HC4-85	0-1.5	weathered shale	9.9	6.8
HC4-85	6.0-7.0	massive shale	22.1	11.1

Highly fractured zones which were described in the core logs appeared to occur in the bedrock at depth. These zones are thought to indicate planes of movement, possibly resulting from post-glacial deformation. K.Y. Lo, (1978) in his discussion of regional distribution of in-situ horizontal stresses in rocks of Southern Ontario, states "inference from case histories, together with direct measurements of in-situ stresses, indicate that high horizontal stresses exist in Silurian and Ordovician (Queenston Formation) rocks". By removing the overlying material, the underlying

rock is deprived of some of its vertical stabilizing support, providing therefore, a triggering mechanism for the failure of the weaker low carbonate content zones. The layer fails by buckling upwards and sliding along the weakest horizontal plane. This release of strain energy accompanying the buckling failure produces substantial heave (Lo, 1978).

Although little quarrying of the shales has occurred on Site F, pre-glacial erosion or the natural phenomenon of deep gullying due to post-glacial erosion may have caused the release of in-situ rock stresses and subsequent fracturing.

Personal communication with Dr. O. White (Ontario Geological Survey, June 9, 1986) confirmed the presence of stress relief ridges associated with the Queenston Formation. It was indicated that if a substantial amount of overburden has been removed by erosional processes, a sufficient amount of stress is present within the rock to form post-glacial deformations.

Review of Ontario Geological Survey Map P.2605 of the Quaternary geology of the Hamilton area, indicates the presence of a pattern of bedrock pressure release ridges which trend along the old Lake Iroquois shoreline. Extrapolation of a zone of stress relief represented by these

surface features follows a strike which could include the Site F location.

To summarize, the available information obtained for the Queenston Formation from the literature, personal discussions and most importantly, the field data, resulted in the interpretation of four zones within the shale:

1) An uppermost weathered or fractured shale.

2) This zone is underlain by a more massive weakly fractured calcium cemented shale.

3) At the lower contact of the massive shale, a fractured pressure release zone appears to occur.

4) This is underlain by a moderately fractured shale which has undergone less calcium cementation.

Figure 4 (see Page 49) is a conceptualized diagram of the Site F geology and hydrostratigraphy.

4.4 Shale Resources

The shale on Site F is similar in texture and chemical composition to the shale mined and used for brick in Waterdown and Tansley (See Table 3).

Twenty percent of the shale at Site F is green (See Table 4) and typically the green shale is reported to be harder due to higher calcium carbonate ($CaCO_3$) content. The chemical tests indicate a slightly higher lime content in the green shale (which may or may not be representative of $CaCO_3$). The higher carbonate content results in a lighter coloured brick.

Small amounts of gypsum were encountered in vugs and along some fracture surfaces at various intervals in Boreholes HC3, HC4, HC5, and HC7. In Borehole HC6, gypsum was noted in one fracture and in Borehole HC8 vugs from which the gypsum was presumably dissolved, were noted at two locations. Gypsum was not observed in Boreholes HC1 and HC2. The vugs were usually in the order of 6 mm in diameter and the veins about 2 to 3 mm thick.

TABLE 3

SUMMARY OF CHEMICAL TESTS
QUEENSTON SHALE

	RED SHALE			GREEN SHALE				COMPOSITION* OF QUEENSTON FORMATION	
	HC3 (8.2m)	HC5 (8.3m)	HC7 (7.2m)	HC3 (8.7m)	HC5 (10.5m)	HC7 (9.1m)	Test Pit Sample	Waterdown	Tansley
SiO_2	46.0	49.7	48.6	46.6	44.4	47.3	44.8	52.0	45.1
Al_2O_3	13.3	12.1	13.3	12.3	8.3	14.0	11.7	14.5	11.9
Fe_2O_3	5.6	5.4	5.7	3.9	2.5	4.3	3.5	6.6	4.6
CaO	12.2	11.8	11.0	14.0	20.1	12.0	15.6	6.8	14.3
MgO	3.3	2.9	3.1	3.1	2.6	3.2	3.2	3.8	2.6
Na_2O	0.1	0.4	0.2	0.2	0.5	0.01	0.2	0.3	0.8
K_2O	4.2	3.7	4.1	3.8	2.6	4.4	3.7	4.1	3.0
TiO_2	0.7	0.8	0.7	0.7	0.6	0.7	0.7	0.8	0.7

*From Table 34, Guillet (1967)

TABLE 4

ESTIMATED PERCENTAGE OF GREEN SHALE

Drilling Sites	Estimated Red Shale	Estimated Green Shale
HC1	85%	15%
HC2	89%	11%
HC3	77%	23%
HC4	77%	23%
HC5	84%	16%
HC6	77%	23%
HC7	82%	18%
HC8	80%	20%
Average	**80%**	**20%**

RESERVES

Reserves of shale were estimated for the licenced portion of the site and for the whole site. For this purpose, it was assumed that working faces would be about 10 m high and benches would be about 3 m wide. Groundwater was not considered a problem since this could be removed by pumping. All of the site was included except for the most northerly portion where the overburden is over 10 m thick.

For the licenced portion of the site, the reserves of shale are estimated to be about 7.5 million cubic metres or about 18 million tonnes, and for the whole site, the reserves of shale are estimated to be about 20 million cubic metres or 48 million tonnes. A quarry depth of about 30 m (the limit of our boreholes) was assumed for the purpose of these calculations.

5.0 SURFACE WATER

5.1 Regional Drainage

Site F located about 2.5 km north of Hamilton Harbour, is situated almost entirely on the west portion of the Indian Creek basin. The northern most edge and the southwestern most part of the site drain into the Falcon Creek watershed (see Drawing 4).

The headwaters of Indian and Falcon Creeks originate on the Niagara Escarpment. Below the escarpment, the creeks meander through a predominantly residential area and ultimately drain into Hamilton Harbour.

It should be noted also that the Rambro and Hager drainage systems have been diverted into the Indian Creek watershed about 1 km above the mouth of Indian Creek.

5.2 Local Drainage

Temporary surface water monitoring stations were established at five culverts along the North Service Road at the south end of the site to monitor stream flow and sample surface water for general chemistry. Locations of the monitoring stations are shown on Drawing 4. The measured discharges are shown in Appendix 7.

High flows resulting from snowmelt runoff were measured during early spring. In April 1986, measured discharge

ranged from 12.3 L/s at Station 3 (Indian Creek) to 0.65 L/s at Station 4 (Indian Creek). Significant stream flows were also noted in November 1985, when discharges of about 104.4 L/s were measured at Station 1 (Falcon Creek) and about 4.2 L/s at Stations 1A and 4 (Falcon Creek). These flows are believed to be higher than normal discharges owing to abnormally high rainfall during November, 1985.

do we have this report?

According to the 1985 Burlington Landfill Monitoring Report (Gartner Lee Associates Limited) there was generally no flow in Falcon and Indian Creeks at the North Service Road during August, September and October. Hydrology Consultants discharge data (See Appendix 7) were similar.

Longer-term (5 years) stream flow data from the hydrometric station (#02HB012) on Grindstone Creek near Aldershot indicate a mean annual discharge of 764 L/s and a low flow of 48.1 L/s. Due to the proximity of Grindstone Creek and its similar basin characteristics, the above data was taken to be representative of the creeks in the study area, and was used to calculate base flow in the hydrologic budget.

is this still true?

A review of the Ministry of the Environment's files on permits to take water indicate that there are no major surface water users in the vicinity of Site F.

5.3 Chemistry

Surface water samples were collected from the five surface water sampling stations (Drawing 4) and were analyzed for major anions and cations, heavy metals, dissolved organic carbon, hardness, pH and phenols. Chemical results are shown in Appendix 12. In addition, the specific conductance of the water in Indian Creek was measured at various locations between Site F and Lake Ontario. These measurements which were made on three occasions are shown in Drawing 4.

Elevated chloride and conductivity at surface water monitoring stations 1A (Falcon Creek) and 3 (the Indian Creek Tributary south of the Bayview Landfill at the North Service Road) indicate the presence of diluted leachate in local surface waters. For example, the chloride concentration measured in the tributary south of the Bayview Landfill is considerably higher than concentrations measured in other Indian Creek tributaries (430 mg/L vs 60 mg/L).

how dilute is the leachate?

Elevated contaminant indicators (measured by the MOE and Hydrology Consultants, 1985) such as conductivity, chloride, phenolics and BOD suggest that the leachate from the Bayview Landfill is degrading surface water in Indian Creek to at least the CN Railway. It is noted that the water quality in Indian Creek improves south of Plains Road.

what significance?

6.0 GROUNDWATER

6.1 Occurrence and Use

Drawing 2 shows the location of known water supply wells in the study area. Most groundwater supplies are obtained from drilled wells or bored wells which penetrate the shale or limestone bedrock (see Tables 5 and 6).

TABLE 5

WATER WELL TYPE

Bored or Dug Wells	Drilled Wells	Unknown Wells	Total
8	47	5	60
13%	79%	8%	100%

TABLE 6

WATER BEARING FORMATION

Bedrock Wells	Overburden Wells	Unknown Wells	Total
44	3	13	60
73%	5%	22%	100%

Water bearing formations in the bedrock usually constitute the source of groundwater supply. According to well records, bedrock wells penetrate about 8 to 69 m into the shale. On average, bedrock wells penetrate 20 m into the bedrock. According to water well records, water yields from bedrock wells range from 1.5 L/m to 54 L/m with an average

of 14 L/m. Almost all bedrock wells identified are located north of Highway 403 on the scarpe face of the Niagara Escarpment or above it.

In the study area, only 5 percent of the residents are known to utilize overburden deposits for water supply. Overburden well depths range from 11 to 13 m with an average depth of 12 m.

No residential well users were identified downgradient of Site F as a result of a review of the well records. However, a resident of the Aldershot area, Mr. Chumley, reported the presence of six residential wells and aided in their identification. Upon further surveys in the area, a total of 13 wells were identified or reported, at distances of more than 1 km from the site. Of these, 3 were identified as being currently used for human consumption. Two others were used only for limited outdoor use on a seasonal basis, and the households were all on municipal supply. The remaining wells inspected were filled in or not in use. In most instances, municipal services supply drinking water to the residents of Aldershot.

6.2 Hydrostratigraphy

Five major hydrostratigraphic units characterize the hydrogeology of Site F. These are illustrated in Figure 4. From youngest to oldest they are:

1) Overburden
   a) Till Unit at north end of site
   b) Discharge Zone Till
2) Fractured Shale Unit - highly weathered at surface
3) Weakly Fractured Massive Shale Unit
4) Highly Fractured Shale Unit
5) Moderately Fractured Bulk Shale Unit

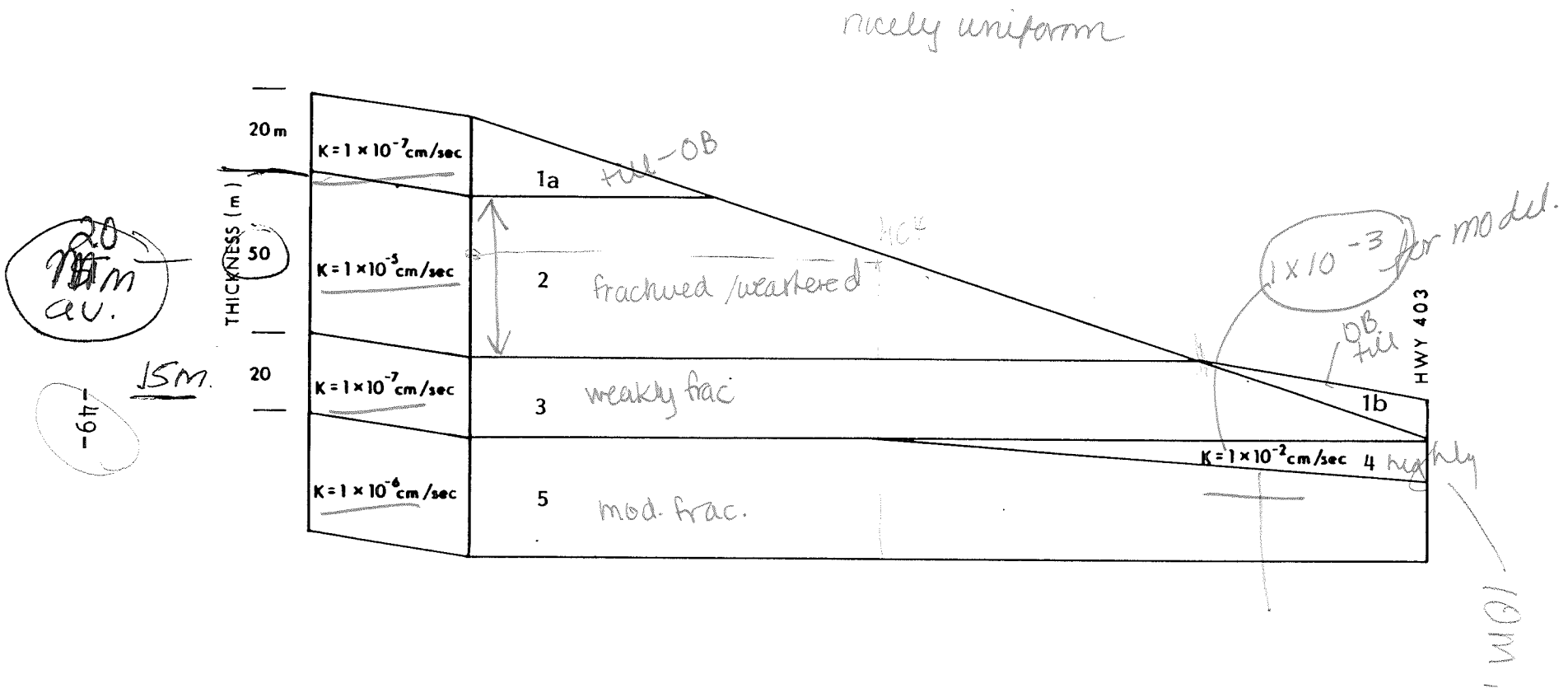
6.2.1 Overburden

a) Till Unit

This unit consists mainly of clayey silt till to silt till. Including data obtained by Gartner Lee Associates Limited from the adjacent Regional Landfill site, the hydraulic conductivities of this till at Site F varied from 5×10^{-8} to 1.7×10^{-7} cm/s. The mean hydraulic conductivity was determined to be 1×10^{-7} cm/s. The effective porosity of this unit was estimated to be 25 percent.

FIGURE 4

SUMMARY OF HYDROSTRATIGRAPHIC UNITS



1) OVERBURDEN
   a) TILL UNIT
   b) DISCHARGE ZONE TILL
2) FRACTURED SHALE UNIT - HIGHLY WEATHERED AT SURFACE
3) WEAKLY FRACTURED MASSIVE SHALE UNIT
4) HIGHLY FRACTURED SHALE UNIT
5) MODERATELY FRACTURED BULK SHALE UNIT

Trow

An upper weathered or fractured zone in the Upper Till is interpreted to extend from ground surface to an approximate depth of 4 to 5 m. Freeze and Cherry (1979) note that deposits of clayey or silty till in Southern Ontario contain networks of hairline fractures which are essentially vertical and which are responsible for hydraulic conductivities one to three orders of magnitude greater than in unfractured materials. Assuming this, the weathered zone is considered to have a hydraulic conductivity of 1×10^{-5} cm/s which is about two orders of magnitude greater than the unfractured Upper Till. This is a very conservative assumption since recent research (Dusseault, 1985) indicates that fractures do not appear to substantially increase the bulk hydraulic conductivity of the till.

what significance? do we agree or not?

The effective porosity of the fractured Upper Till is considered to be 0.001.

do we agree

OK.

b) Discharge Zone Till

South of Site F, the till is overlain by lacustrine sediments, mostly sand. This granular layer is typically a black deposit grading toward the Lake from a thin veneer of sands to sandy silts near Site F.

An average hydraulic conductivity of 3×10^{-6} cm/s was assigned to the combined lacustrine sands and clayey silt till overburden south of Site F. This is based on

calculated hydraulic conductivities of the overburden unit south of Site F and is representative of the combination of sands and till.

6.2.2 Fractured Weathered Shale Unit

This unit represents the upper section of the shale bedrock. Hydraulic conductivities measured in the upper bedrock zone ranged from 1 x 10^{-6} cm/s to 1 x 10^{-4} cm/s with a geometric mean of 1 x 10^{-5} cm/s. This hydraulic conductivity is considered representative of the weathered shale. The effective porosity is considered to be about 0.001.

This unit is observed in the deeply incised on-site gullies, and contributes shallow groundwater discharge to the intermittent streams that occupy the gullies.

6.2.3 Weakly Fractured Massive Shale Unit

The massive shale unit is a calcite cemented shale with a low frequency of joints and fractures. The hydraulic conductivity of this unit is demonstrated by the packer injection tests conducted on HC6-85. Zones of low hydraulic conductivity (10^{-7} cm/s to 10^{-8} cm/s) were indicated by essentially zero water loss into packed-off intervals of the shale formation. Zones of very low hydraulic conductivity, as low as 10^{-10} cm/s, were also reported by other investigators (Hewetson, 1985).

The weakly fractured massive shale unit is a complex unit in that it occurs throughout the investigated shale sequence and exhibits variable, but generally low hydraulic conductivities. For this reason, a hydraulic conductivity value of 1 x 10^{-7} cm/s is considered to be conservative but representative.

This value provides the necessary permeability contrast among the shale units to produce vertical groundwater gradients which in most cases are similar to those measured on-site.

6.2.4 Highly Fractured Shale Unit

A highly fractured zone has been delineated from the interpretation of core logs and pumping tests. The zone appears to be a wedge of fractured shale, underlying the weakly fractured massive unit. It is interpreted to extend from the midpoint of the landfill site to about 500 m south of Highway 403, where it is overlain by overburden.

Pumping test No. 1 measured the hydrogeologic characteristics of this unit. The test was run for 24 hours by Northern Well Drilling Ltd. to measure the aquifer parameters and investigate boundary conditions of the highly fractured shale. Drawdowns were recorded in observation wells and piezometer nests at OW1, MB12, MB16 and WAT7. Time drawdown vs plots were analyzed using the Jacob semi-

log method to calculate transmissivity, storativity, and hydraulic conductivity. A geometric mean of the values calculated from this analysis are shown in Table 7.

TABLE 7

HYDRAULIC CHARACTERISTICS OF THE HIGHLY FRACTURED SHALE

	Geometric Mean
Transmissivity, m^2/sec	4.1×10^{-5}
Storativity	5.7×10^{-5}
Hydraulic Conductivity, cm/s	1.3×10^{-2}

The distance vs drawdown diagram for this pump test indicates that at t = 1000 minutes, the extent of the drawdown cone would be 180 m or greater from the pumping well. Curves of drawdown vs distance and drawdown vs time are presented in Appendix 10.

6.2.5 Moderately Fractured Bulk Shale Unit

This unit represents the remaining bulk of the Queenston Formation to the depth investigated during this study. It is considered to be moderately fractured and has a mean hydraulic conductivity of about 1×10^{-6} cm/s.

6.2.6 Other Fracture Zones

Two other pumping tests were conducted on-site which resulted in drawdowns at distance from the pumping well. PW2, 20 m from HC3-85, showed hydraulic connection between

the wells. The duration of the test and its constant pumping rate of 2.5×10^{-4} m^3/s indicated a relatively extensive fracture zone in the weathered shale (see Appendix 10).

PW3 was pumped at a low discharge rate of 3.0×10^{-5} m^3/sec to maintain suction during the pumping test. This fracture zone showed connection with observation wells at WAT5 but the amount of interference was small. The projected cone of influence from the distance-drawdown diagram suggested that the cone of influence for this test would extend about 22 m from the pumping well (see Appendix 10). This test result indicates a fracture zone within the weathered shale that has limited lateral extent.

6.3 Bedrock Groundwater Flow System

6.3.1 Regional

Drawing 13 shows the regional bedrock potentiometric contour map. The map represents the groundwater flow regime in the shallow bedrock. Groundwater movement in the shale bedrock is generally southeasterly towards Hamilton Harbour. The horizontal gradient of the shallow bedrock flow system between Site F and Lake Ontario varies from 0.009 to 0.02.

The data also indicate a deep groundwater flow system in the bedrock that probably originates above the Niagara Escarpment north of Site F. This flow system moves under Site F at depth (greater than 30 m) and discharges in the vicinity of Lake Ontario.

6.3.2 Local

Drawing 15 shows the local bedrock potentiometric contour map. The horizontal gradient of the shallow bedrock flow system on-site ranged from 0.01 to 0.05. The direction of groundwater movement is southeasterly. The groundwater contours indicate the potential for groundwater discharge in the shallow weathered shale to discharge on-site into the deeply incised gullies.

6.4 Overburden Groundwater Flow System

Groundwater elevations measured in April, 1986, from standpipe installations in the overburden were used to generate the regional and local watertable contour maps.

6.4.1 Regional

The watertable slopes southeasterly towards Hamilton Harbour (see Drawing 14). The horizontal gradient of the shallow flow system between Site F and Hamilton Harbour varied from 0.009 to 0.02. Streams between Site F and Hamilton Harbour act as discharge zones for the shallow groundwater flow system. It is interpreted that interstream zones between

Site F and Lake Ontario are localized recharge areas that discharge to these streams.

6.4.2 Local

Drawing 16 represents the watertable configuration at Site F. Generally the watertable slopes southeasterly. In the vicinity of deeply incised gullies, groundwater flow paths are controlled by local discharge areas on Site F.

Depth to the watertable varied from about 0.15 m below ground surface at HC8-85 to about 2.9 m below ground surface at HC7-85. The horizontal gradient of this shallow flow system 0.03 to 0.045 on Site F.

6.5 Distribution of Vertical Groundwater Gradients

The data on vertical groundwater gradients identify strong downward gradients on Site F and generally lateral to weak upward gradients between drilling sites HC9-85 and HC14-85 (south of Site F) and (HC13-85) near Hamilton Harbour. Typical measured vertical groundwater gradients are summarized in Table 8.

TABLE 8

VERTICAL GROUNDWATER GRADIENTS
APRIL, 1986

Geology	Site F Range	Between Site F and Hamilton Harbour Range
Between overburden and shallow weathered shale	0.01-1.1 (downward gradient)	0.002-0.1 (upward gradient)
Between shallow weathered shale and deep moderately fractured bulk shale	0.1-1.1 (downward gradient)	0.008-0.09 (downward gradient)

It is important to note the vertical groundwater gradients between Site F and Hamilton Harbour. The vertical gradients between the overburden and the shallow weathered shale are weakly upwards while the vertical gradients between shallow weathered shale and the deep shale formation are downwards. This supports the interpretation that the shallow groundwater flow system represents active local zones of recharge and discharge, while the regional groundwater flow system is active at depth and does not discharge between Site F and Hamilton Harbour.

6.6 Computer Model of Groundwater Flow System

6.6.1 General

Computer model studies were undertaken to provide a more detailed analysis and understanding of the existing groundwater flow system at Site F and between Site F and Hamilton Harbour, in terms of the hydrostratigraphy, flow

paths, infiltration/discharge occurrences and rates, and potential contaminant migration paths.

6.6.2 Model Description

A steady-state, two-dimensional, cross-sectional groundwater flow model was applied to the regional study area. The framework of the model is based on the following equations:

(Potentials)

$$\frac{\partial}{\partial x} K_x \frac{\partial h}{\partial x} + \frac{\partial}{\partial y} K_y \frac{\partial h}{\partial y} = 0 \qquad (1)$$

and

(Streamfunctions)

$$\frac{\partial}{\partial x} \frac{1}{K_y} \frac{\partial \psi}{\partial x} + \frac{\partial}{\partial y} \frac{1}{K_x} \frac{\partial \psi}{\partial y} = 0 \qquad (2)$$

where

x	=	horizontal coordinate axis
y	=	vertical coordinate axis
K_x	=	hydraulic conductivity in the x-direction
K_y	=	hydraulic conductivity in the y-direction
h	=	hydraulic head or potential
ψ	=	streamfunction or flow line.

Equations (1) and (2) describe the steady-state, two-dimensional flow of groundwater in porous media.

A finite element solution to these equations was developed by Frind and Matanga (1985), and presented in the form of a computer program called FLONET. This program solves for the two-dimensional (a) potential (hydraulic head) distribution; and (b) flow line (streamfunction) distribution within a porous media.

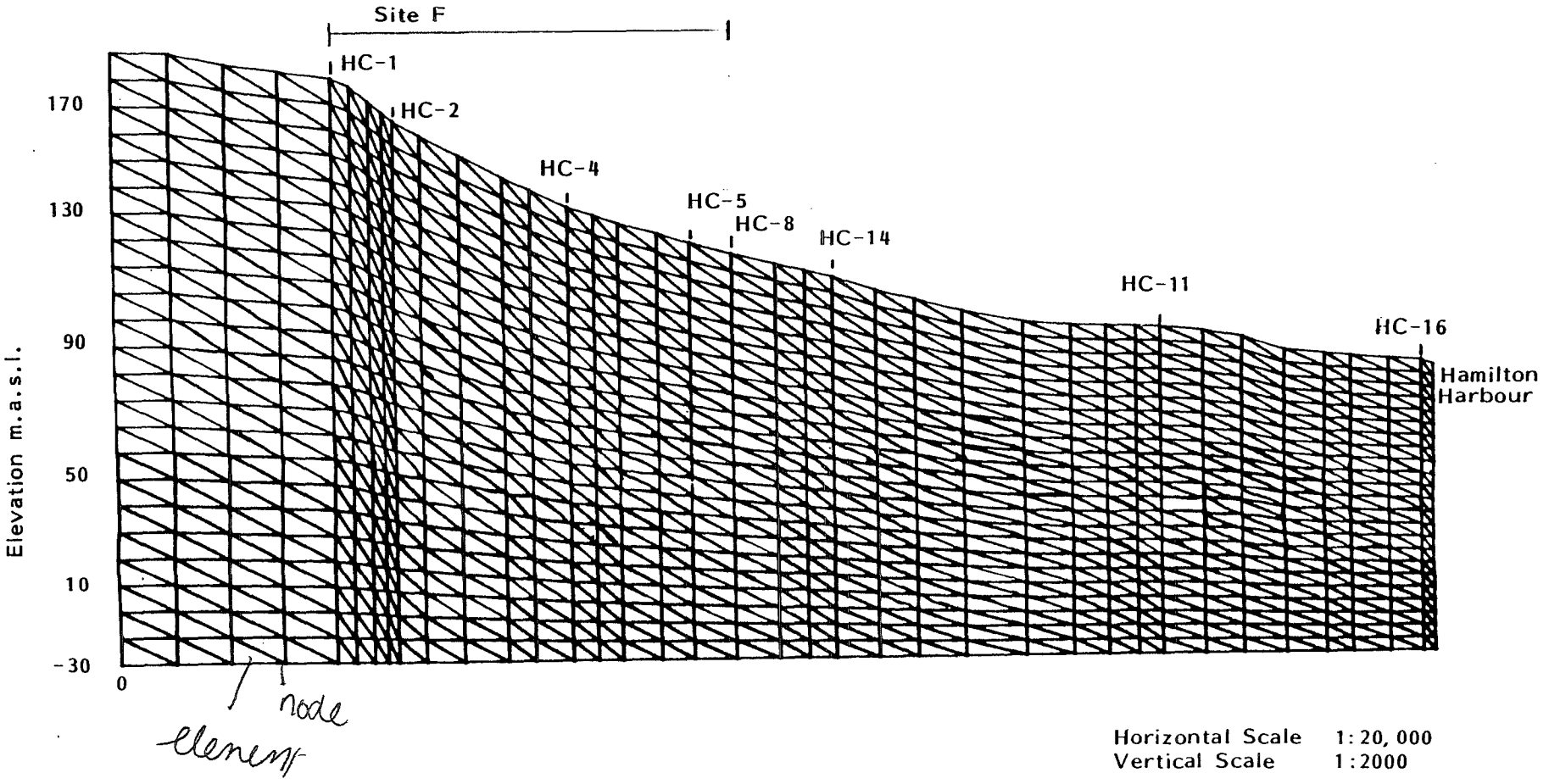
At Site F, the bulk of the near surface geological deposits are shale in which the hydraulic conductivity is controlled to a large extent by fractures. It is not practical to model the discrete fracture flow system at this site. The data required to attempt such an exercise is complex and difficult to obtain. Therefore, a simplified and reasonable approach was adopted whereby the fractured porous media at Site F are represented by an equivalent non-fractured porous media. It is also assumed that conditions do not change appreciably (i.e., in the lateral dimension) across the site.

6.6.3 Model Calibration

The model represents a vertical cross-section of a line passing through the centre of Site F, from a starting point about 800 metres north of the site, to Hamilton Harbour (see Section B-B, Drawing 2). The finite element grid consists of 912 nodes and 1702 elements (see Figure 5).

FIGURE 5

FINITE ELEMENT GRID



Trow

The upper boundary (water table) of the model is defined as a fixed head boundary corresponding to water levels measured in April 1986, in several observation wells located along the line of the cross-section. The left hand or upgradient side, and the base of the model are defined as no-flow boundaries. These boundaries represent a single flow line. The right or downgradient side of the cross-section is defined as a fixed head boundary corresponding to the elevation head of Lake Ontario.

The hydrostratigraphic units described in Section 6.2 were the basis of the computer simulation. The final model calibration was accomplished after more than 50 simulations, which included analyzing the effects of variable boundary conditions, heterogeneity, and to a lesser extent anisotropy. The resulting computer simulation represents a reasonable approximation of the field data with respect to geology, hydraulic conductivity, water table, and groundwater gradients (horizontal and vertical).

do we agree with the simulation?

6.6.4 Discussion

The calibrated, steady-state groundwater flow net is shown in Figure 6. The flow lines illustrate that groundwater infiltration (or recharge) occurs from the north (left) boundary of the model, across and beyond the proposed landfill site, to the vicinity of drilling site HC14-85, south of Highway 403, roughly 500 metres beyond the southern

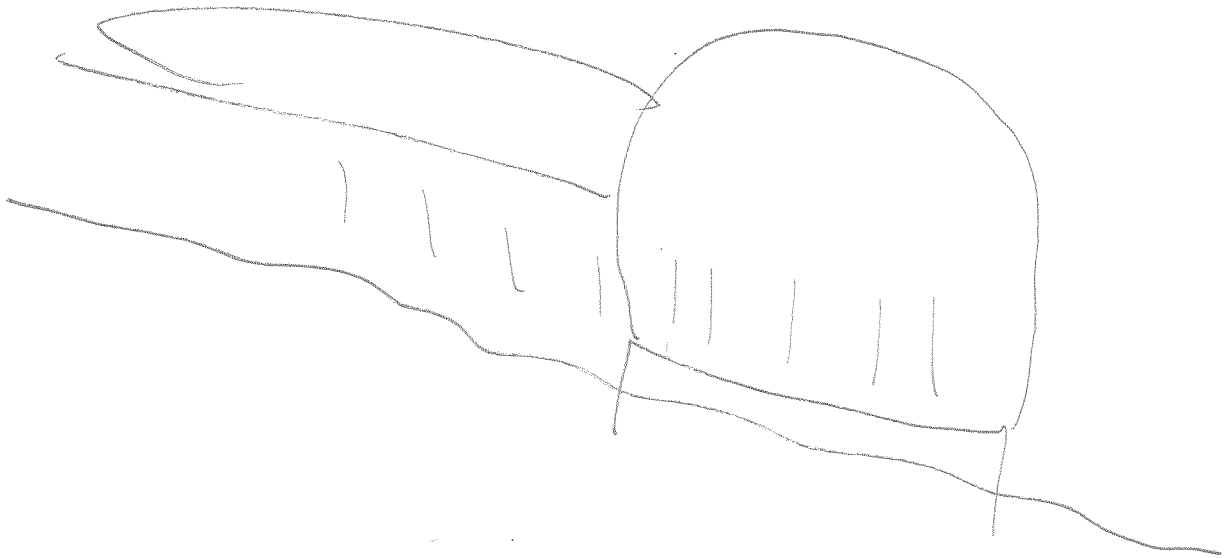
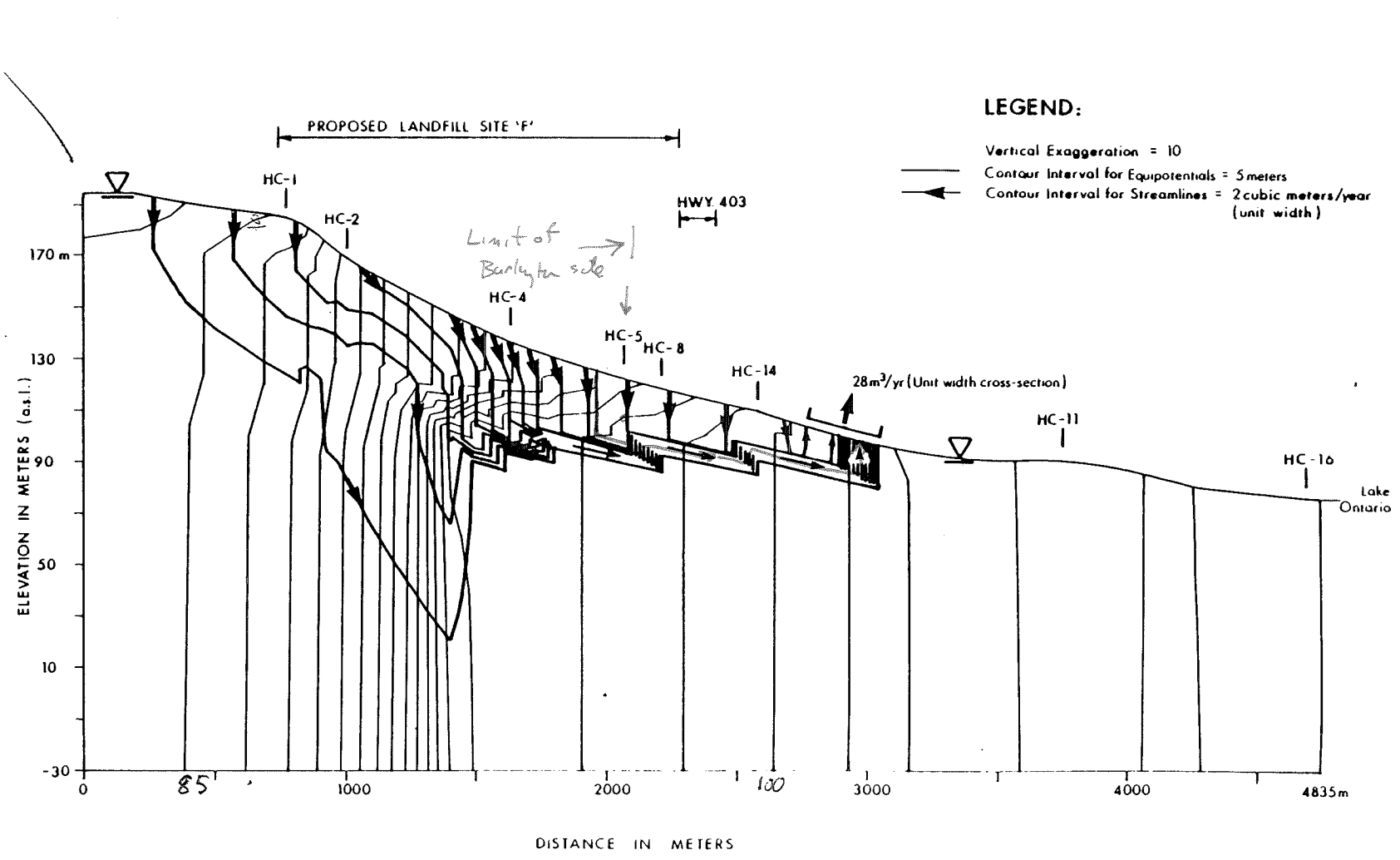


FIGURE 6

COMPUTER SIMULATION OF GROUNDWATER FLOW SYSTEM



border of the landfill. Much of the recharge is indicated to occur over the area of the landfill itself, where the average infiltration rate is calculated to be about 13.7 mm/year, and the total volume of infiltrating water (using a site width of 500 metres) is in the order of 10,000 m^3/year (4.2 Imperial gallons per minute). The regional groundwater flow system is represented by the flow line originating north of Site F and discharging to Lake Ontario.

The highly fractured shale unit acts as a permeable underdrain to the system. Most of the groundwater that infiltrates over the proposed landfill site (and some from beyond) moves horizontally through this unit, and discharges at surface where the highly fractured unit is interpreted to outcrop beneath the glacial till, south of Highway 403. No groundwater which infiltrates over the landfill area is shown to discharge between this zone (noted above) and Lake Ontario.

Consequently, the highly fractured shale unit appears to exert a major control on the groundwater flow system, and in combination with the overlying weakly fractured massive shale unit, results in the large downward vertical gradients recorded at depth across and beyond the landfill site.

With respect to contaminant migration, the model indicates that leachate contaminated groundwater discharge (via the highly fractured shale unit) will be limited to a small area (62 acres) south of HC9-85 and HC14-85 (south of Hwy 403). The total discharge rate would be about 14,000 m^3/year (6 gpm).

7.0 GROUNDWATER CHEMISTRY

7.1 General

Thirty observation wells at drilling sites HC1-85 to HC13-85 were sampled and analyzed in order to establish the ambient groundwater chemistry of the various strata at Site F. Samples were collected from the wells in January, 1986, and were submitted to Barringer Magenta Ltd. (Rexdale) for analysis. The chemical results are documented in Appendix 12. The analysis included major anions, cations, metals, hardness, dissolved organic carbon, phenols and pH.

In order to define the chemical variations in the overburden, water chemistry profiles of anions and cations were prepared (see Drawings 18 and 19) and chemical results were plotted on a Piper trilinear diagram (see Drawing 17). Drawing 21 shows the chloride concentration distribution in the study area.

7.2 Background Chemistry

As groundwater moves through the saturated zone, an increase of total dissolved solids normally occurs. The composition of the medium through which the groundwater moves through and the length of time the groundwater has been associated with this medium is generally reflected by the dominant anionic species.

Table 9 is an interpretation of the dominant ionic species at Site F. The results of the Piper analysis generally indicate a dominant sulphate facies in the upper hydrostratigraphic units and a chloride dominant facies in the lower hydrostratigraphic units. Off-site, (between HC9-85 and Hamilton Harbour) chloride facies seem prominent both near surface and at depth. Chloride concentrations exceeding 10,000 mg/L were measured. This distribution of dominant hydrochemical facies would indicate a slightly less mature flow system near surface at Site F underlain by a more mature geochemical facies which becomes dominant both near surface and at depth off site.

In order to assess the potential of the shales to be a source for direct diffusion of chlorides, cores were analyzed at HC7, HC8 and HC12 for natural chloride content. Appendix 3(iv) shows the results of this analysis. The chemical data shows no correlation between in-situ shale and groundwater chloride concentration.

The in-situ concentrations of chloride within the groundwaters sampled throughout the study area exhibit high concentrations of chloride mainly as a result of their mature chemical nature.

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TABLE 9

GEOCHEMICAL FACIES

Drilling Site	OW	Cations	Anions
HC1-85	1	Magnesium	Sulphate
	2	Magnesium	Sulphate
	3	Magnesium	Sulphate
HC4-85	1	Calcium Magnesium	Chloride
	2	Calcium Magnesium	Sulphate
	3	Calcium	Sulphate
	4	Calcium	Sulphate
HC5-85	1	Sodium	Chloride
	2	Sodium Calcium	Sulphate
	3	Magnesium Calcium	Sulphate
	4	Magnesium Calcium	Sulphate
HC8-85	1	Sodium	Chloride
	2	Sodium	Chloride
	3	Sodium	Chloride
	4	Sodium	Chloride
HC9-85	1	Sodium	Chloride
	2	Sodium	Chloride
	3	Magnesium Calcium	Sulphate
	4	Calcium Magnesium	Sulphate
HC10-85	1	Sodium	Chloride
	2	Sodium	Chloride
	3	Calcium Sodium	Bicarbonate Sulphate
HC11-85	1	Sodium	Chloride
	2	Sodium	Chloride
HC12-85	1	Sodium	Chloride
	2	Sodium	Chloride
	3	Sodium Calcium	Chloride
HC13-85	1	Sodium	Chloride
	2	Sodium	Chloride
	3	Calcium	Chloride

7.3 Analysis of Organic Chemistry

Preliminary chemical data obtained from Gartner Lee Associates Limited in connection with the plume delineation study at the Regional landfill site in Burlington (Burlington Landfill site) show no off-site migration of priority pollutant organic compounds.

The 1985 monitoring report on the adjacent Burlington Landfill site prepared by Gartner Lee Associates Limited, indicate that leachate attenuation and dilution appear to renovate leachate quality sufficiently that it is indistinguishable from background water quality in the shale. They conclude that it is unlikely that the leachate plume will ever cause any significant off-site impact on groundwater.

Hydrology Consultants sampled observation wells installed at HC15-86, 600 m downgradient of the Regional Landfill (Burlington Landfill) site. Waters from these wells at depths of about 6 m, 12 m and 28 m showed no detectable concentrations of priority pollutant organic compounds (see Appendix 12).

According to investigations by Hewetson (1985), the occurrence of TCA (1,1,1-trichloroethane) at MB18 approximately 875 m south of the Bayview Landfill site is evidence of groundwater contamination related to the landfill that commenced operation about 30 years ago.

It should be noted that the TCA concentrations at MB-18 could only be detected in parts per trillion.

Groundwater sampled from a well installed at the bedrock contact at drilling site HC9-85, (near MB-18), about 875 m downgradient of the Bayview Landfill site, showed no detectable concentrations (in parts per billion) of priority pollutant organic compounds (see Appendix 12).

7.4 Tritium Analysis

7.4.1 General

Tritium (^3H), a radioactive isotope of hydrogen, enters the hydrological cycle from both man-made and natural sources.

Naturally occurring tritium results from the interaction of cosmic-ray-produced neutrons with nitrogen in the upper atmosphere. Prior to 1952, the natural tritium content of precipitation was estimated to be between 5 and 20 T.U. (T.U. is a Tritium unit and one Tritium unit = one ^3H atom in 10^{18} atoms of hydrogen). The half-life of tritium is approximately 12.3 years, and so groundwaters that were recharged with naturally-enriched tritium (prior to 1952), are expected to have present tritium concentrations below 2-4 T.U.

Beginning in 1952, the above ground testing of thermonuclear bombs resulted in an increase in the ^3H concentration in the

atmosphere. Post-1952 precipitation contains a greater ^{3}H concentration (in 100's of T.U.) and so this isotope can be used to distinguish pre-1952 waters from post-1952 waters.

Brown (1961) has established 15.3 T.U. as being a representative tritium level for pre-1952 atmospheric waters. In the 34 years which elapsed between 1952 and the sampling date, the naturally-occurring tritium will have undergone decay corresponding to 2.8 half-lives. Therefore, pre-1952 atmospheric waters are calculated as yielding a present day activity of approximately 2.3 T.U. This value is used in the present study to differentiate between groundwaters recharged prior to 1952 and those waters infiltrating after 1952.

Tritium is a conservative tracer in groundwater studies since it is not significantly affected by reactions other than radioactive decay (Freeze & Cherry, 1979).

7.4.2 Sampling

A total of eight water samples were collected from observation wells installed at drilling sites HC5-85 and HC8-85. The samples were sent to the University of Miami's School of Marine and Atmospheric Sciences for enriched tritium analysis. Each drilling site consisted of four wells screened at approximate depths of 3, 10, 17 and 30 meters. The surficial clay till is about 8 m thick at

HC5-85 and about 5.5 m thick at HC8-85. Shale bedrock underlies the till at both locations.

The results for all sampling locations are given in Appendix 12 (iii).

7.4.3 Overburden

The tritium content in the clay till ranges from a low of 1.03 T.U. at HC8-85 to a high of 36.61 T.U. at HC5-85. If a value of 2.3 T.U. is used to delineate pre-1952 groundwater from post-1952 groundwater, the results indicate that bomb tritium has fully penetrated the till unit at HC5-85 (8 m) whereas a value of 1.03 T.U. in the shallow well at HC8-85 indicates penetration has not reached 3 metres.

This difference is likely due to spatial variability in the hydraulic conductivity or dessication in the upper clay till.

7.4.5 Bedrock

Tritium concentrations in the shale range from a low of 0.21 T.U. at HC8-85 (9.8 m) to a high of 9 T.U. at HC5-85 (17.4 m). The absence of bomb tritium in the shale at HC8-85 is likely attributable to the low concentrations found in the overlying till at this location. It is also evident that the zones monitored by wells at HC8-85 are not hydraulically connected to fractured shale that may have

been recharged since 1952. The tritium concentrations of 4.57 and 9.0 in HC5-1 and HC5-2 respectively are relatively low, however, could possibly represent groundwater younger than 1952. It is possible that fractures in the shale at these depths are hydraulically connected to a surface as a result of the highly fractured nature of the overlying till discussed previously.

While the results indicate that there is a moderate degree of variability in the depth of penetration of post 1952 water, the data suggests a general decrease in tritium concentrations locally. Previous tritium samples taken near the Bayview Landfill site by Hewetson, et al (1985) yielded somewhat inconclusive results regarding the vertical distribution of ^{3}H. While many more locations were sampled, much of the data indicated increasing concentrations of tritium with depth. This could be a result of a very complex network fractures or from bias introduced during the monitor installation or sampling phase.

The 1986 data indicates a general decrease in tritium concentration with depth and a variable depth of ^{3}H penetration over the site as a result of local variations in vertical fracturing.

8.0 IMPACT ASSESSMENT

8.1 Water Balance

The general Hydrologic Budget of a basin can be quantified by using the equation.

$$P = E + Qs + R + \Delta S$$

where P = Precipitation

E = Evapotranspiration

Qs = Surface Runoff

R = Groundwater Recharge

Δ S = Change in Storage

Climatological data were obtained for the Burlington area from the Atmospheric Environmental Service of Environment Canada. Annual norms of precipitation and evapotranspiration based on 10 year averages found in Appendix 6.

An average of 869 mm of precipitation falls on the Burlington area per year, of which about 511 mm is returned to the atmosphere by evapotranspirational processes.

Since long-term averages were used in the water balance, changes in storage in the hydrologic system are negligible. Therefore, groundwater recharge and surface runoff amount to 869 - 511 mm = 358 mm per year.

8.2 Groundwater Contaminant Migration

8.2.1 Introduction

The assessment of the environmental impact of leachate from a municipal waste landfill on groundwater requires an adequate understanding of the migration of potential contaminants in the groundwater. Mechanisms including advection (flow of groundwater), diffusion and hydrodynamic dispersion will act to transport contaminants in groundwater. At the same time, mechanisms such as adsorption, precipitation, volatilization, biodegradation and matrix diffusion will act to attenuate or retard the transport of contaminants. The following sections describe transport and attenuation mechanisms and relate their importance to the evaluation of groundwater contaminant migration.

8.2.2 Transport Mechanisms

8.2.2.1 Advection

Advection is the movement of contaminants in groundwater due to the flow of groundwater through the subsurface. For contaminants which do not adsorb significantly on the subsurface soils or rock (non-reactive contaminants) or are not significantly removed from the groundwater by precipitation, volatilization or biodegradation reactions (conservative contaminants), migration occurs at the rate of the average linear groundwater velocity.

Groundwater flow through soil or rock is typically described using Darcy's Law which relates the groundwater flux, v (volume of groundwater flow per unit area per unit time, i.e., - $m^3/m^2/day$) to the hydraulic conductivity, K and the hydraulic gradient, i.

Darcy's Law is stated as:

$$v = Ki \qquad (1)$$

The average linear groundwater velocity is a key parameter in the evaluation of contaminant migration in groundwater. Groundwater velocities can range from several hundred metres per year in sand and gravel or fractured rock, to less than several centimetres per year in unfractured clay or crystalline rock. The average linear velocity V, is related to the Darcy flux v, and the effective porosity of the soil or rock, n by:

$$V = v/n \qquad (2)$$

Effective porosity is the proportion of the pore space through which groundwater actually moves. The effective porosity in soils ranges from 10 percent to 50 percent and represents the bulk of the intergranular pore size. However, in fractured or fissured materials the bulk of the groundwater flow occurs through the fractures. The effective porosity of the fractures expressed as a percent

of the rock mass can be extremely small, in the range of 0.1 percent to 0.001 percent as estimated based on hydraulic conductivity-fracture opening size-fracture frequency relationships (see Hoek and Bray, 1977, p. 133). This very small effective porosity can result in high calculated linear groundwater velocities in fractured rock despite relatively low hydraulic conductivities.

8.2.2.2 Diffusion

In groundwater environments in which groundwater velocities are very low, such as in unfractured rock or clay, chemical diffusion can be the dominant mechanism of contaminant transport. Diffusion is a chemical process by which dissolved compounds move through the groundwater due to concentration gradients rather than moving due to the groundwater flow. Diffusion will occur from zones of high concentration to zones of lower concentration. The flux rate of contaminants, F (mass of contaminant per unit area per unit time, i.e., $mg/m^2/day$) is related to the total porosity, n, the effective diffusion coefficient, De, and the concentration gradient, i_c by:

$$F = n\ De\ i_c \qquad (3)$$

The effective diffusion coefficient is a measure of the material's ability to allow chemical diffusion through the water filling its pores. Diffusion coefficients for various

geologic materials vary over a much smaller range than hydraulic conductivity. The value of the effective diffusion coefficient tends to decrease with decreasing porosity. Diffusion coefficients for non-reactive dissolved contaminants range from 5×10^{-6} cm^2/s in clay materials with a porosity of 40 percent (Desaulniers, 1984) to 10^{-8} cm^2/s in shales and limestones (J.A. Cherry, personal communication).

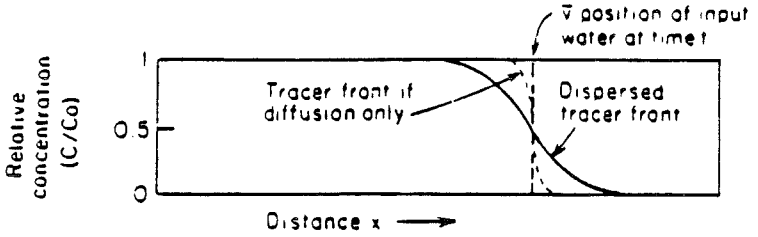
Because of the nature of diffusive contaminant transport it is not possible to refer to an actual contaminant velocity. Diffusive flux and therefore velocity varying in time and space depending on the concentration gradient. Based on typical diffusion coefficient values and the transport scales of interest (i.e, metres), diffusive transport can account for contaminant velocities of centimetres per year at the most and will only be important when advective transport velocity is significantly lower than centimetres per year.

8.2.2.3 Dispersion

Dispersion is a secondary contaminant transport mechanism which can contribute to the spreading of contaminants in groundwater (see Figure 8). Due to heterogeneities in the geologic media, the flow paths followed by the groundwater contaminants can vary. Because of this, a spreading of the contaminant occurs. This spreading is predominantly in the

FIGURE 8

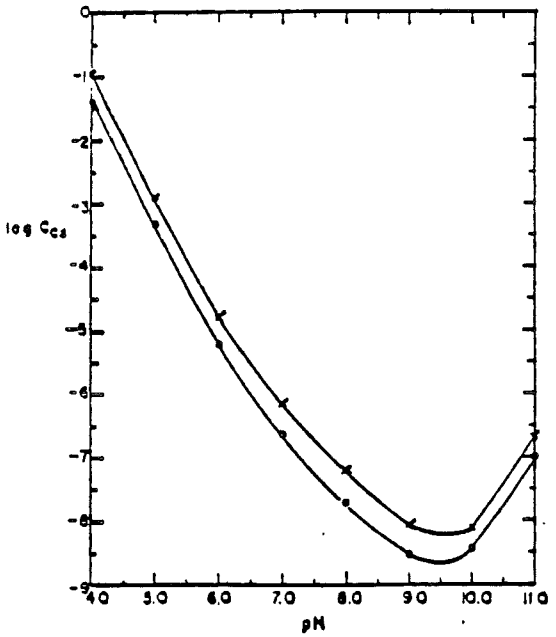
ILLUSTRATION OF THE EFFECT OF DISPERSION (Cherry, 1979)



Schematic diagram showing the contribution of molecular diffusion and mechanical dispersion to the spread of a concentration front in a column with a step-function input.

FIGURE 9

EXAMPLE OF THE EFFECT OF pH ON CADMIUM SOLUBILITY (Hem, 1972)



Solubility of cadmium as a function of *p*H. Total dissolved carbon dioxide species concentration, 10^{-2} moles/l. Ionic strengths of 0.00 (circles) and 0.10 (crosses).

longitudinal direction (along the groundwater flow path) although dispersion also occurs in transverse directions. Dispersion is a mechanism that, does not have a clear theoretical basis. The traditional approach to describing dispersion in groundwater relates the coefficient of hydrodynamic dispersion, D, to the dynamic dispersivity, a, the average linear groundwater velocity, V, and the coefficient of molecular diffusion D* or nDe (see equation 3) by:

$$D = aV + D^* \qquad (4)$$

From this relationship it is evident that the degree of dispersion increases with increasing groundwater velocity. The diffusion term is negligible except for very low groundwater velocities.

8.2.3 Contaminant Attenuation Mechanisms

8.2.3.1 Adsorption

Inorganics

Adsorption is a process by which contaminants can be removed from the groundwater and bound onto the surface as soil and rock particles. This removal of contaminants from the groundwater reduces contaminant concentrations and results in a slowing or attenuation of contaminant migration. Adsorption of cationic (positively charged) inorganic contaminants occurs predominantly on clay minerals and iron oxyhydroxide coatings on mineral grains.

In cation exchange, some cation species are preferred on exchange sites on the soil or rock surface. In general, cations with higher ionic charges and smaller ionic radii are preferred. These tendencies are illustrated by the following series:

Strongly Adsorbed Weakly Adsorbed

$$Cs^+ > K^+ > Na^+ > Li^+$$

$$Ba^{2+} > Sr^{2+} > Ca^{2+} > Mg^{2+}$$

Divalent (2+) cations are more strongly adsorbed than monovalent cations. The majority of heavy metal cations which are of environmental concern are small divalent cations and are generally strongly adsorbed.

For a given contaminant in a given groundwater and soil, the degree of contaminant adsorption is described by the distribution coefficient, K_d, which is the ratio of the concentration of contaminant adsorbed on the soil to the equilibrium concentration in the groundwater. K_d is typically determined by laboratory tests. The higher the K_d the more strongly the contaminant is adsorbed.

Organics

The adsorption of organic contaminants from groundwater on soil and rock occurs primarily by preferential partitioning of the contaminants on solid natural organic matter. For

some organic compounds adsorption may also occur on clay minerals or other soil particles.

Based on laboratory testing of soils and sediments, researchers such as Karickhoff et al. (1979) have been able to relate the K_d of organic contaminants in soils to the organic carbon content of the soil, f_{oc}, and a partition coefficient, K_{oc}, describing the degree of contaminant adsorption on the organic matter.

$$K_d = (K_{oc})(f_{oc}) \qquad (5)$$

K_{oc} can be determined by laboratory experimentation for particular contaminants on particular types of organic matter. However, empirical relationships have been developed which describe the K_{oc} for organic contaminants in terms of its water solubility, S, or octanol/water partition coefficient K_{ow}.

Values for solubility and K_{ow} for most organic compounds are readily available in the scientific literature. Therefore, with information on the soil organic matter content and estimates of the K_{oc}, K_d values for most organic parameters can be estimated.

Retardation due to Adsorption

The effect of adsorption reactions on contaminant migration is typically expressed by the retardation equation:

$$R = V/V_c = 1 + (P_b/n)K_d \quad (6)$$

where R is the retardation factor, V is the average linear groundwater velocity, V_c is the velocity of the contaminant, P_b is the bulk density of the soil and n is the porosity. The contaminant velocity decreases or retardation factor increases as the K_d increases. In fractured rock with a non-porous matrix, adsorption will occur only on the fracture surfaces. The effect of adsorption reactions in fractured rock is typically expressed by:

$$R = 1 + 2K_a/b \quad (7)$$

where b is the fracture aperture and K_a is the ratio of adsorbed contaminant on a per unit surface area basis to the concentration in the groundwater. K_a is similar to K_d and is also determined by laboratory experiments.

8.2.3.2 Precipitation

Precipitation reactions in the groundwater system can be an effective attenuation mechanism for many heavy metal contaminants. Precipitation of inorganic contaminants as insoluble mineral phases can remove the contaminants from the groundwater. Precipitation reactions will be controlled by the geochemical conditions in the groundwater.

In general, precipitation reactions which result in the attenuation of inorganic contaminants such as heavy metals will occur predominantly in association with other geochemical changes. Increases in pH as a result of other reactions will result in the precipitation of many heavy metals which form insoluble hydroxides or carbonates. Similarly, heavy metals made mobile by the presence of organic compounds which act as complexing agents, can be made immobile due to precipitation when the organic complexing agents are destroyed by biodegradation in the groundwater system.

The potential for precipitation reactions can be evaluated qualitatively using theoretical solubility calculations and geochemical relationships. These relationships are frequently summarized graphically by means of concentration versus pH diagrams such as shown in Figure 9 (see page 80). These relationships generally consider only simple systems with two or three components and do not accord to the large number of geochemical reactions/interactions which can occur. More complex intereactions can be evaluated using computerized geochemical equilibrium models.

8.2.3.3 Biodegradation and Microbial Transformations

Biodegradation is a process by which dissolved organic contaminants in groundwater are destroyed by the action of micro-organisms (generally bacteria) which occur naturally

in the subsurface. Biodegradation can be a very effective mechanism for the attenuation of organic contaminants. Biodegradation can occur as bacteria utilize the contaminants as a food/energy source. The rate of biodegradation for various organic contaminants is highly variable depending on the type of contaminant, the bacterial populations present and the general geochemical conditions in the groundwater.

8.2.3.4 Matrix Diffusion

Matrix diffusion can be an important attenuation process which can control the migration of contaminants through fractured or fissured geologic materials. In fractured rocks and fissured clays, the bulk of the groundwater flow generally occurs through the fractures because the relative permeability of the fractures can be significantly higher than that of the rock or clay matrix between the fractures. The bulk of the contaminant transport will also occur through the fractures. Although the rock or clay matrix has a very low ability to transmit water, the matrix can still have a substantial water-filled porosity. Sedimentary rocks such as limestones and shales can have porosities of 1 to 10 percent and clays can have porosities of up to 40 percent. As contaminated groundwater flows through the fractures, there will be a chemical concentration gradient between the contaminated water in the fracture and the uncontaminated water in the matrix. This concentration gradient will result in diffusion of contaminants from the fracture into

the matrix. In this way, contaminants are removed from the groundwater flowing through the fracture and the apparent contaminant migration along the fractures will be restricted. The principle of matrix diffusion as an attenuation process is illustrated in Figure 10.

Matrix diffusion coefficients and matrix porosities can be measured by means of laboratory tests. Fracture groundwater flow velocities can be estimated from hydraulic conductivity, hydraulic gradient and fracture porosity relationships. Using this information, the effect of matrix diffusion on contaminant migration for simple fracture geometrics can be evaluated by means of computer models such as those developed by Tang et al. (1981) and Sudicky and Frind (1982).

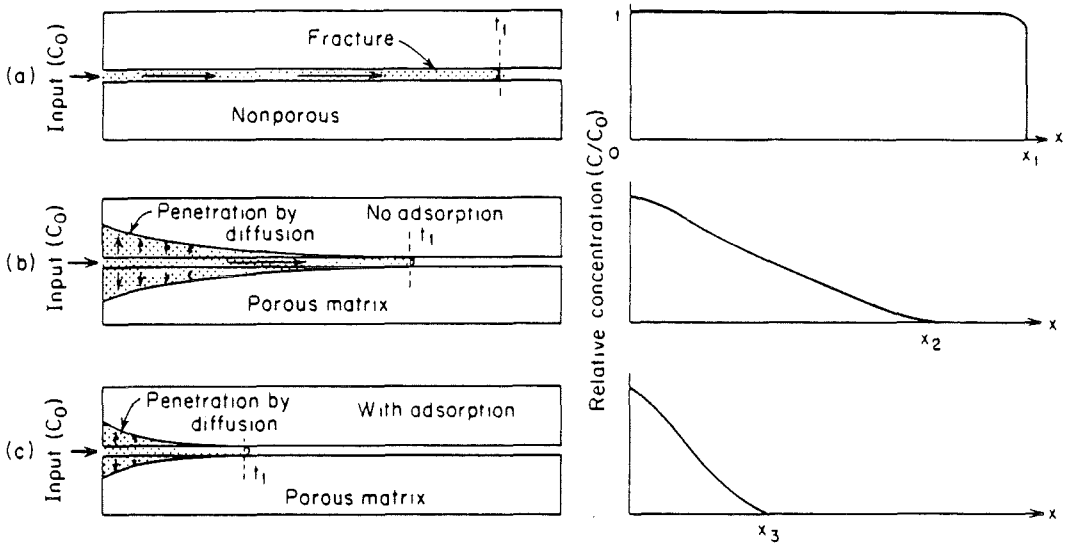
8.3 Municipal Landfill Leachate

8.3.1 Chemical Composition

Leachate from municipal landfill sites forms as a result of percolation of precipitation or groundwater through the refuse in the site. The percolating waters acquire dissolved inorganic and organic contaminants by leaching of soluble materials and by decomposition of organic matter in the refuse. The chemical composition of landfill leachate can vary with time during landfill operation and following site closure as the contaminant is leached. Similarly, the chemical composition of leachate can vary within a site or between sites as a result of variations in the types of refuse in the sites.

FIGURE 10

ILLUSTRATION OF THE EFFECT OF MATRIX DIFFUSION
(Cherry, 1979)



Effect of diffusion on contaminant migration in porous fractured medium. (a) Unidirectional hydraulic transport in a fracture in a nonporous medium; (b) unidirectional hydraulic transport with migration into matrix as a result of molecular diffusion; (c) unidirectional hydraulic transport with molecular diffusion and adsorption (profiles of relative concentration of reactive contaminant within fracture shown at time t_1).

Municipal landfill leachates are typically characterized by elevated concentrations of ions such as calcium, sodium, sulfate, chloride, bicarbonate and ammonia. Metals such as iron and manganese frequently occur at elevated concentrations. Concentrations of heavy metals such as copper, nickel, lead and zinc are not generally elevated in municipal landfills except in cases associated with the disposal of industrial wastes. The presence of organic contaminants in landfill leachates is indicated by high concentrations of organic carbon, organic nitrogen and chemical oxygen demand (COD). Many of the toxic organic chemicals on the United States Environmental Protection Agency (US EPA) List of Priority Pollutants have been identified in landfill leachates. In general, however, the largest proportion of the organic compounds in landfill leachate are non-toxic but still undesirable contaminants such as carboxylic acids resulting from the decomposition of organic matter in the municipal refuse.

The chemical composition of landfill leachate and the variability in the composition of landfill leachate can be illustrated by a study on Landfill Leachate and Treatability done for the Regional Municipality of Halton by Pollutech Limited in 1985-1986. As part of this study, leachate from the Regional Landfill sites in Burlington, Georgetown and Oakville were analyzed for complete inorganic and organic characterization. The results of the inorganic analyses are

shown in Table 10 and the organic analyses are shown in Table 11.

The principal constituents of the leachates from the three Halton Region Landfills are inorganic compounds such as calcium, magnesium, sodium, ammonia and alkalinity. The pH of the landfill leachates is nearly neutral. Concentrations of heavy metals such as arsenic, barium, chromium, mercury and zinc are relatively low and are generally below Ontario Ministry of the Environment Drinking Water Objectives. The overall inorganic chemical compositions of the leachates are relatively similar.

A variety of organic chemicals are present in the landfill leachates. These chemicals include both volatile and semi-volatile organic chemicals, many of which are on the US Environmental Protection Agency (EPA) List of Priority Pollutants. Low concentrations of these organic chemicals in the several to several hundred microgram per litre range (ug/L) have also been identified by Reinhard et al. (1984) at municipal landfills in the Region of Waterloo and in North Bay, Ontario. The occurrence of these organic compounds in leachate should be considered typical for municipal landfills. However, the concentrations of volatile organic chemicals such as 1,1,1-trichloroethane, 1,1-dichloroethylene, 1,2-dichloroethane, diethyl ether, methyl ethyl ketone and xylene greater than 1000 ug/L which

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TABLE 10

INORGANIC CONSTITUENTS OF LANDFILL LEACHATE - HALTON REGION (ANALYSIS BY POLLUTECH LTD.)

	Burlington Landfill	Georgetown Landfill	Oakville Landfill
GENERAL PARAMETERS			
pH	6.7	7.0	7.0
Total Solids	7600	1900	4000
Acidity (as CaCO3)	400	72	96
Alkalinity (as CaCO3)	4000	1500	2500
Chloride	1200	370	940
Chemical Oxygen Demand (COD)	5300	200	500
Biochemical Oxygen Demand (BOD5)	4300	26	55
Ammonia (as N)	350	120	170
Kjeldahl Nitrogen (as N)	330	140	190
Nitrate (as N)	2.9	1.0	1.2
Total Phosphorus	0.8	ND	0.82
Soluble Phosphorus	0.3	ND	0.1
Cyanide	0.006	0.004	0.007
METALS			
Aluminum	0.61	ND	ND
Antimony	0.0053	0.0011	0.0041
Arsenic	0.0077	0.0059	0.0041
Barium	1.4	0.9	0.9
Boron	6.2	3.1	5.2
Calcium	570	220	240
Chromium	0.08	0.05	0.03
Iron	110	29	28
Magnesium	400	94	220
Managanese	7.9	1.6	1.4
Mercury	0.002	0.00005	0.00073
Nickel	0.11	0.03	0.07
Silica	14	12	14
Sodium	790	310	680
Zinc	4.6	ND	0.3

All concentrations in mg/L.

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TABLE 11

PRINCIPAL ORGANIC CONSTITUENTS OF LANDFILL LEACHATE - HALTON REGION (ANALYSIS BY POLLUTECH LTD.)

	Burlington Landfill	Georgetown Landfill	Oakville Landfill
VOLATILE US EPA PRIORITY POLLUTANTS (ug/L)			
1,1,1-Trichloroethane	1400	<1	<1
1,1,2,2-Tetrachloroethylene	220	<0.5	<0.5
1,1-Dichloroethylene	4200	2.1	11
1,2-Dichloroethane	3000	8.0	<1
Benzene	150	3.2	<1
Ethyl benzene	400	4.9	<0.5
Freon II	52	3.6	3.6
Methylene chloride	390	0.9	1.7
Toluene	890	2.7	<0.5
Trichloroethylene	940	2.5	1.2
VOLATILE NON-PRIORITY POLLUTANTS (ug/L)			
Camphor	67	4.7	<1
Diethyl ether	1300	27	69
Dimethyl sulfide	1100	<2	<2
Fenchone	80	4.2	8.4
Methyl ethyl ketone	2000	6.7	11
Styrene	81	<1.5	<1.5
Xylene	5200	90	<1.5
SEMI-VOLATILE US EPA PRIORITY POLLUTANTS (ug/L)			
Dimethyl phthalate	11	<2	<2
Diethyl phthalate	21	5.0	<2
Butylbenzyl phthalate	18	<1.5	<1.5
Bis(2-ethylhexyl) phthalate	6.0	200	<1.5
SEMI-VOLATILE NON-PRIORITY POLLUTANTS (ug/L)			
Cresols	170	<10	<10
ORGANIC ACIDS (mg/L)			
Acetic	43	NA	NA
Propionic	130	8.4	NA
Isobutyric	43	2.5	NA
Butyric	230	14	NA
Isovaleric	91	5.7	NA
Hexanoic (2 isomers)	310	15	NA
Heptanoic	11	<1	NA
Octanoic	11	NA	NA
Phenylacetic	5.3	<1	NA

NA- Not analysed.

occur in the Regional Landfill in Burlington leachate are not typical of municipal landfills. These organic chemicals are commonly used as industrial solvents and the chemical data suggest that the Burlington Landfill has received significant quantities of industrial wastes for disposal.

8.3.2 Relative Mobility of Groundwater Contaminants

The various inorganic and organic contaminants which can be present in landfill leachate will not have the same degree of mobility in groundwater due to the varying effect of attenuation processes for different contaminants. A contaminant such as chloride is highly mobile in groundwater because it is virtually unaffected by geochemical attenuation processes such as adsorption, precipitation and biodegradation. There are, however, few such contaminants which are highly mobile under all conditions. Most inorganic and organic contaminants in groundwater are relatively mobile in some situations and immobile in others. For example, most heavy metal contaminants in groundwater are relatively immobile due to adsorption and precipitation reactions except under conditions of low pH (<6) or in the presence of significant complexing agents in the groundwater or leachate. Similarly, organic contaminants such as trichloroethylene are generally relatively mobile in groundwater except in settings where the geological materials have significant concentrations of solid organic matter on to which the trichloroethylene is readily

adsorbed. Whether a particular contaminant is mobile or immobile will depend primarily on the characteristics of the geological materials and the chemical composition of the groundwater or leachate. In the context of a municipal landfill, several generalizations can be made with regard to the mobility of contaminants.

Contaminant	General Mobility
Chloride	Highly mobile not affected by geochemical attenuation processes.
Major cations: Calcium, Magnesium, Potassium, Sodium	Mobile but attenuated to some degree by adsorption and precipitation reactions.
Ammonia	Mobile but attenuated to some degree by adsorption and by oxidation to nitrate in aerobic environments.
Sulfate	Mobile but attenuated by precipitation reactions and by reduction to sulfide in anaerobic environments.
Heavy metals: Cadmium, Chromium, Copper, Lead, Mercury, Nickel, Zinc	Immobile except under conditions of low pH (< 6) or in presence of complexing agents.
Arsenic, Selenium	Expected to be mobile based on presence as anionic species but not often observed to be mobile.
Boron	Expected to be mobile but behaviour is poorly known.
Iron, Manganese	Mobile but attenuated by adsorption and oxidation-precipitation reactions.

Contaminant	General Mobility
Volatile Organic Chemicals	Mobile but highly variable in degree of mobility. Adsorption and biodegradation are key attenuation mechanisms.
Semi-volatile Organic Chemicals	Less mobile than volatiles but highly variable in degree of mobility. Adsorption and biodegradation are key attenuation mechanisms.
Organic Acids	Mobile but susceptible to attentuation by biodegradation.

The potential degree of variability in the mobility of volatile and semi-volatile organic chemicals in groundwater can be illustrated by examining the K_{OC} of the organic chemicals. The K_{OC} will indicate the affinity of the chemicals to adsorb on solid soil organic matter and the extent to which the chemicals can be attenuated by adsorption. The table in Appendix 13 shows the volatile and semi-volatile organic chemicals which were identified in the Burlington Landfill leachate together with their log K_{OC} values. In a given geologic material, chemicals with low K_{OC} values (less than 2.0) such as diethyl ether, methyl ethyl ketone, methylene chloride, 1,1-dichloroethylene and 1,2-dichloroethane will be the most mobile of the organic chemicals. Chemicals with high Koc values (greater than 5.0) such as butylbenzyl phthalate and bis(2-ethylhexyl) phthalate will be the most immobile of the organic chemicals.

8.4 Factors Affecting Contaminant Migration

8.4.1 Advection

Advection will be the principal mechanism for the transport of contaminants in the groundwater at Site F. Transport of contaminants due to groundwater flow will be more significant than transport due to diffusion in all the geologic strata at Site F. Contaminant transport by advection is controlled by the linear groundwater velocity (see Section 8.2.2.1). The linear groundwater velocity is calculated based on the hydraulic conductivity and porosity of the hydrostratigraphic unit and the prevailing hydraulic gradient. Based on the measured and estimated values for these parameters presented in the preceding section, Table 12 summarizes the linear groundwater velocities in the various hydrostratigraphic unit at Site F. Groundwater velocities in the weathered shale and the highly fractured shale are approximately 250 and 3150 m per year respectively. Groundwater velocities through the weakly fractured massive shale and the discharge zone till are considerably lower at approximately 0.3 m per year.

8.4.2 Diffusion

Diffusion will not be a significant mechanism for contaminant transport at Site F. However, for the weathered shale and highly fractured shale, contaminant transport will occur predominantly through fractures. The diffusion of contaminants from the fractures into the porous rock matrix

TABLE 12

SUMMARY OF LINEAR GROUNDWATER VELOCITY

	HYDROSTRATIGRAPHIC UNIT	GROUNDWATER FLOW	HYDRAULIC CONDUCTIVITY (cm/s)	HYDRAULIC GRADIENT (m/m)	EFFECTIVE POROSITY	LINEAR GROUNDWATER VELOCITY (cm/s)	LINEAR GROUNDWATER VELOCITY (m/yr)
#2	Weathered Shale	Downward from landfill	1.0 x 10-5	0.08	0.001	8.0 x 10-4	250
#5	Weakly Fractured Shale	Downward from landfill	1.0 x 10-7	1.0	0.10	1.0 x 10-6	0.3
#4	Highly Fractured Shale	Laterally to the south	1.0 x 10-2	0.01	0.01	1.0 x 10-2	3150 **
#1b	Discharge Zone Till	Upward to ground surface	3.0 x 10-6	0.08	0.25	9.6 x 10-7	0.3

(see Section 8.2.2.2) may significantly attenuate migration through the fractures. Effective diffusion coefficients for the shale at Site F have been determined in the laboratory to be approximately 5 x 10^{-7} cm^2/s (J. Cherry, University of Waterloo, personal communication).

8.4.3 Dispersion

The key parameter influencing the degree of dispersion or spreading of groundwater contaminants is the hydrodynamic dispersivity (see Section 8.2.2.3). The value of the dispersivity in the various hydrostratigraphic units at Site F cannot be measured directly. Based on laboratory and field experimental studies, the longitudinal (in the direction of groundwater flow) dispersivity is regarded to increase with increasing contaminant transport distance and may vary in value from less than 5 percent to 100 percent or more of the transport distance. In general, dispersivity values are larger in more highly heterogeneous strata such as fissured rock.

For this assessment, dispersivity values of 5 percent, 10 percent and 30 percent of the transport distance have been assumed for the clayey till, weakly fractured massive shale and weathered/highly fractured shale, respectively.

8.4.4 Adsorption

Inorganics

Adsorption of inorganic contaminants such as heavy metals will generally be a significant attenuation mechanism in situations where the geologic strata have a sufficient cation exchange capacity (CEC) and the pH of the groundwater is 6 to 7 or higher. The total CEC of the various hydrostratigraphic units at Site F are shown in Appendix 3. These results indicate relatively high values for the CEC of the shale bedrock at about 65 meq/100 gm. CEC values such as these reflect a significant potential for the shale to adsorb inorganic groundwater contaminants.

Adsorption of inorganic contaminants such as heavy metals will be enhanced when pH conditions in the groundwater are 6 to 7 or greater. The carbonate mineral content of the geologic strata will be the principal factor which will control the pH of the groundwater. The neutralization or buffering capacity of the carbonate minerals (calcite and dolomite) will tend to neutralize acidic leachate which may form in landfills and will keep the pH of the groundwater above 6 to 7. The carbonate mineral content of the strata at Site F were measured using the Chittick method (Dreimanis, 1962) and are shown in Appendix 5. Carbonate mineral contents such as these represent a very large neutralization capacity and will maintain the pH of the groundwater at levels which will encourage the adsorption of most metal contaminants.

Organics

The adsorption of organic contaminants on geologic materials will be controlled by the total organic carbon content (foc). The foc contents of the various hydrostratigraphic units at Site F were measured using a sulfuric acid-dichromate digestion method and are shown in Appendix 3. The foc in the shale range from 0.10 to 0.64 weight percent with a mean value about 0.18 weight percent.

8.5 Evaluation of Contaminant Migration

8.5.1 General

Based on the preceding description of the hydrogeological conditions at Site F and measurements and estimates of the factors which will control the attenuation of groundwater contaminants, an evaluation of potential contaminant migration at the site can be conducted.

The patterns of potential contaminant migration through the various hydrostratigraphic units at Site F can be illustrated using a series of one-dimensional analytical models which simulate the effects of advection, hydrodynamic dispersion, diffusion and attenuation due to adsorption. The formulation of these models is based on the Ogata-Banks equation (Bear, 1979. Pg. 268).

$$\frac{C}{Co} = \frac{1}{2}\left[\operatorname{erfc}\frac{x - Vct}{(4Dt)^{0.5}} + \exp\frac{Vcx}{D}\operatorname{erfc}\frac{x + Vct}{(4Dt)^{0.5}}\right] \quad (6)$$

$$D = aV + D^* \quad (7)$$

$$D^* = n\frac{De}{R} \quad (8)$$

$$V_c = \frac{V}{R} \quad (9)$$

where:

C/Co - is the relative concentration at distance x and time t

R - is the retardation factor

V - is the linear groundwater velocity

Vc - is the velocity of the contaminant

D - is the coefficient of hydrodynamic dispersion

a - is the dispersivity

D^* - is the coefficient of molecular diffusion

n - is the effective porosity

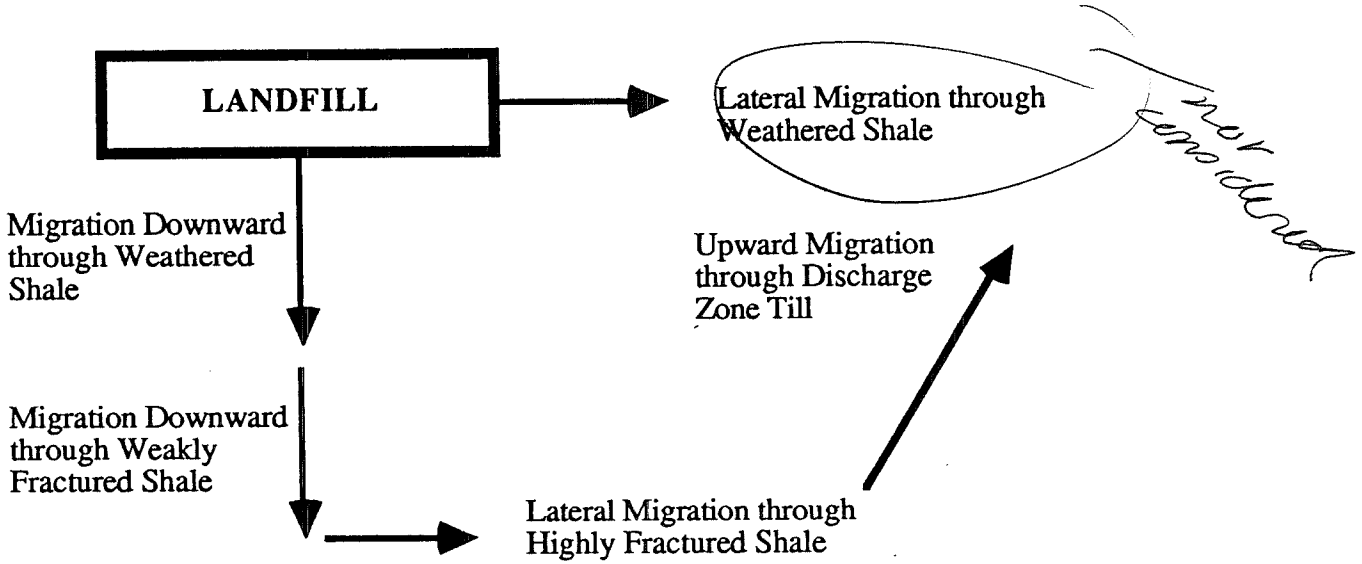
De - is the effective diffusion coefficient

The models assume a constant contaminant concentration input.

Contaminant migration models were developed for four individual components of the groundwater flow system at Site F. The groundwater contaminant pathways at Site F are illustrated in Figure 4. The contaminant migration models

FIGURE 11

SCHEMATIC OF GROUNDWATER CONTAMINANT PATHWAYS



are based on the hydrostratigraphic units described in Section 6.2 (see Figure 4). Lateral migration through the shallow weathered shale was not considered because all such migration would discharge on-site rather than migrate off-site.

Contaminant migration through the weakly fractured massive shale and the discharge zone till considered migration of:

1) A non-adsorbed contaminant (R=1) such as chloride which travels at the same rate as the groundwater.

2) A weakly adsorbed contaminant.

3) A moderately adsorbed contaminant.

Because heavy metal contaminants are expected to be strongly adsorbed and virtually immobile at Site F, the migration of potentially more mobile organic contaminants was considered. Organic compounds identified in the Regional Landfill site in Burlington include weakly adsorbed organic contaminants such as methyl ethyl ketone and methylene chloride which have organic carbon partition coefficient (log Koc) values of less than 1.5.

Typical moderately adsorbed organic contaminants include compounds such as xylene and trichloroethylene which have log Koc values between 2.0 and 3.0.

Using a representative log Koc value for the groups of contaminants and data for the organic carbon content, density and porosity of the various strata, representative retardation factors were calculated using Equation 6. The retardation factors used in the component models are shown in Appendix 14.

The component models for contaminant migration through the weathered shale and the highly fractured shale do not consider adsorption because groundwater flow, and thus contaminant migration is predominantly through fissures. Therefore the groundwater contaminants do not have the opportunity to contact the adsorbing materials in the bulk till of rock matrix and it is not possible to estimate the degree of adsorption on the fissure surfaces.

Matrix diffusion may act to attenuate the migration of all contaminants (non-adsorbed and adsorbed) through the fractures. Matrix diffusion is not considered in the component models.

The results of the contaminant migration simulations are summarized in Figure 12 to 18. The detailed calculations and parameter values are shown in Appendix 15. Later in this report, the component models are coupled together to allow simulation of the complete off-site contaminant migration pathway.

8.5.2 Migration Downward through Weathered Shale

Figure 12 illustrates the C/Co (relative concentration with respect to the input concentration) at the base of the weathered shale beneath the landfill. The first arrival of a non-adsorbed contaminant (R=1), as distinguished by C/Co=0.1, occurs at approximately 10 days. The migration rate through this weathered shale will be very rapid with the input concentrations (C/Co=0.1) reaching the base of the weathered shale after approximately 100 days. Matrix diffusion may act to extend these arrival times but they will likely still be of the order of less than a year.

8.5.3 Migration Downward through Weakly Fractured Massive Shale

Figure 13a and Figure 13b illustrate the C/Co at the base of the weakly fractured shale. There is a small difference between the arrival curves for a non-adsorbed contaminant (R=1) and a weakly adsorbed contaminant (R=1.48) such as methyl ethyl ketone or diethyl ether. The first arrival (C/Co=0.1) at the base of the weakly fractured shale occurs at approximately 30 years. Moderately adsorbed contaminants (R=16) such as xylene or 1,1,1-trichloroethane are significantly retarded with first arrivals (C/Co=0.1) occurring at approximately 500 years.

Trow

FIGURE 12

MIGRATION DOWNWARD THROUGH WEATHERED SHALE

Uncoupled Transient Model - Constant Contaminant Input

Input Parameters:

Downward Groundwater Velocity - ~~8.0~~ x 10^{-4} cm/s
Thickness of Weathered Shale - 20 m
Dispersivity - 6 m

CONTAMINANT CONCENTRATION AT BASE OF WEATHERED SHALE

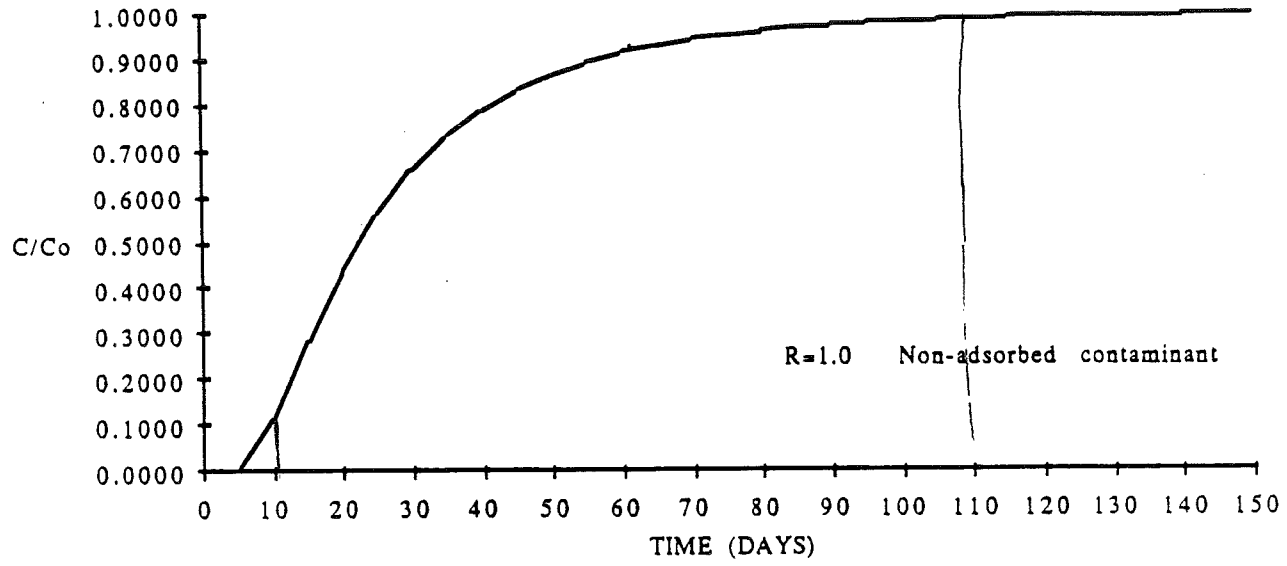


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FIGURE 13a

MIGRATION DOWNWARD THROUGH WEAKLY FRACTURED MASSIVE SHALE

Uncoupled Transient Model - Constant Contaminant Input

Input Parameters:

Downward Groundwater Velocity - 1.0 x 10^{-6} cm/s
Thickness of Weakly Fractured Shale - 15 m
Dispersivity - 0.75 m
Coefficient of Molecular Diffusion (R=1) - 5.0 x 10^{-7} cm^2/s

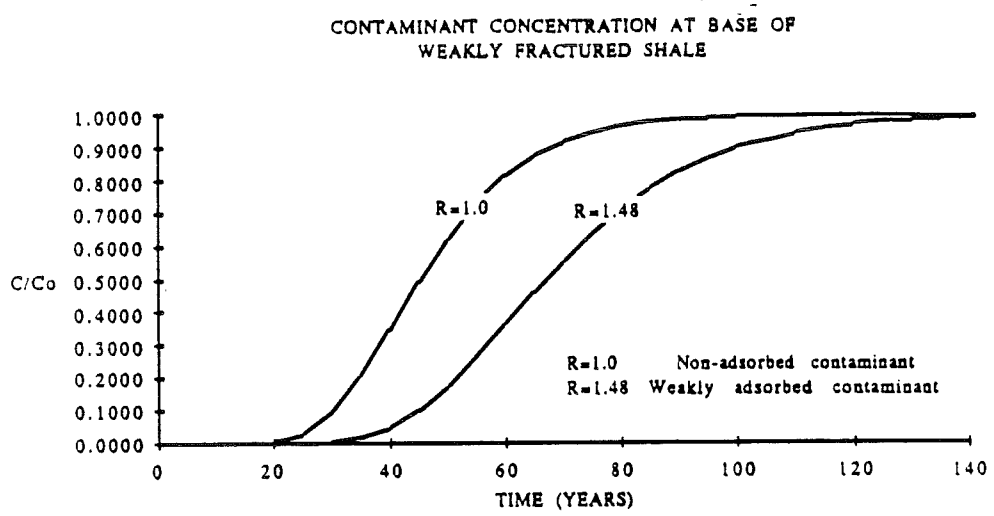


FIGURE 13b

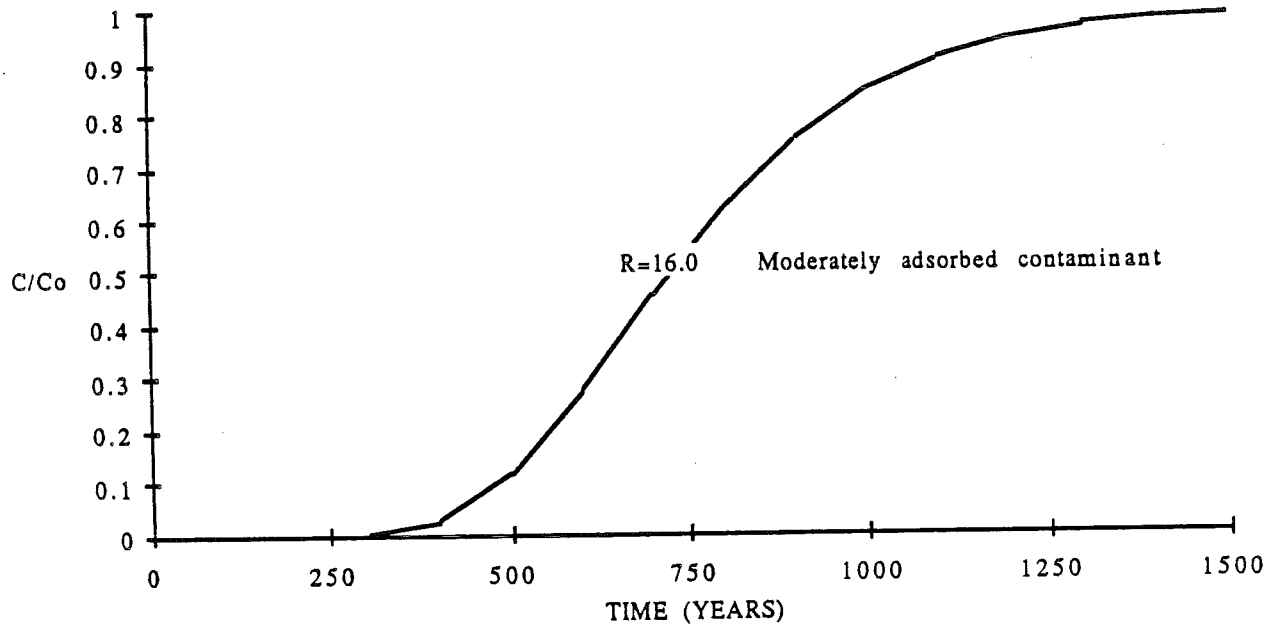
MIGRATION DOWNWARD THROUGH WEAKLY FRACTURED MASSIVE SHALE

Uncoupled Transient Model - Constant Contaminant Input

Input Parameters:

Downward Groundwater Velocity - 1.0×10^{-6} cm/s
Thickness of Weakly Fractured Shale - 15 m
Dispersivity - 0.75 m
Coefficient of Molecular Diffusion (R=1) - 5.0×10^{-7} cm^2/s

RELATIVE CONCENTRATION (C/Co) AT THE BASE OF THE WEAKLY FRACTURED SHALE



8.5.4 Lateral Migration through Highly Fractured Shale

Figure 14 illustrates the C/Co in the highly fractured shale at the discharge zone at a distance of 1400 m from the landfill. For non-adsorbed contaminants, first arrivals (C/Co=0.1) occur at approximately 75 days. Matrix diffusion may act to extend these arrival times but they will likely still be of the order of less than a year. This component model is not coupled with the component models for the weathered shale and weakly fractured shale but does serve to illustrate contaminant migration rates through this unit. Groundwater velocities in the highly fractured shale are extremely rapid and consequently contaminant migration rates are also rapid.

8.5.5 Upward Migration through the Discharge Zone Till

Figure 15 illustrates the C/Co at the ground surface in the discharge zone following upward migration of groundwater through the till overlying the highly fractured shale. For non-adsorbed, weakly adsorbed contaminants there is little difference in the first arrival times. The first arrivals (C/Co=0.1) for the contaminants occur at approximately 25 to 35 years. First arrivals for a moderately adsorbed contaminant are approximately 250 years.

Trow

FIGURE 14

LATERAL MIGRATION THROUGH HIGHLY FRACTURED SHALE

Uncoupled Transient Model - Constant Contaminant Input

Input Parameters:

~~Downward~~ lateral Groundwater Velocity - 1.0 x 10^{-2} cm/s
Distance to Discharge Zone - 1400 m
Dispersivity - 140 m

RELATIVE CONCENTRATION (C/Co) AT THE DISCHARGE ZONE IN THE HIGHLY FRACTURED SHALE

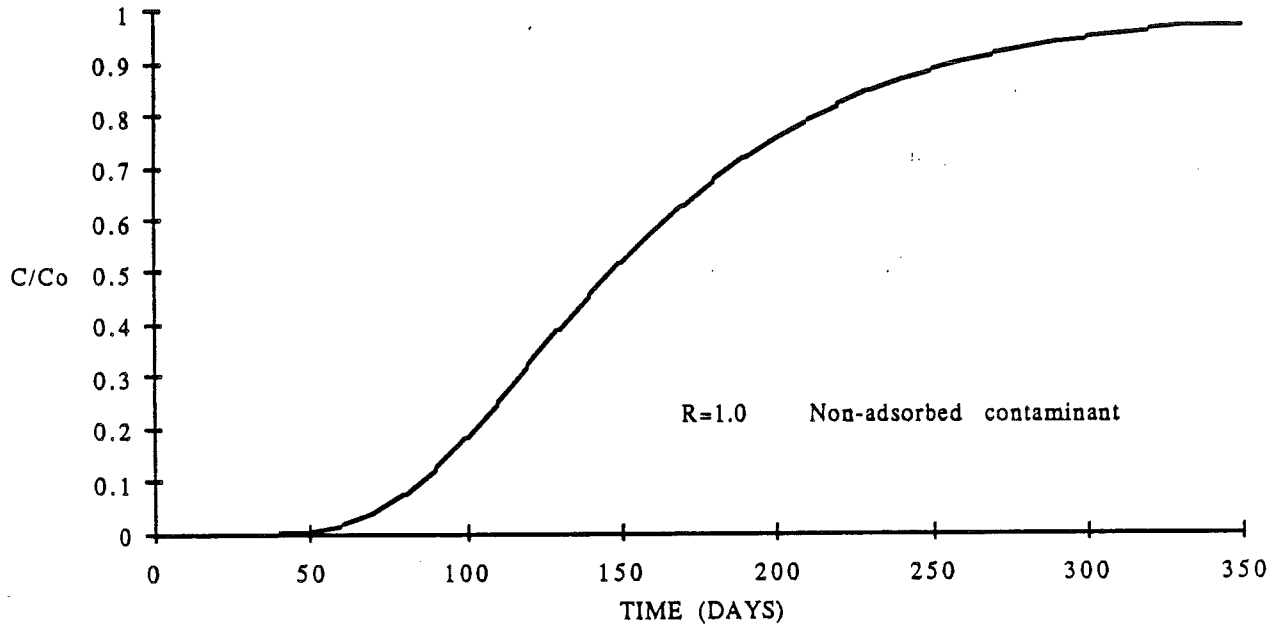


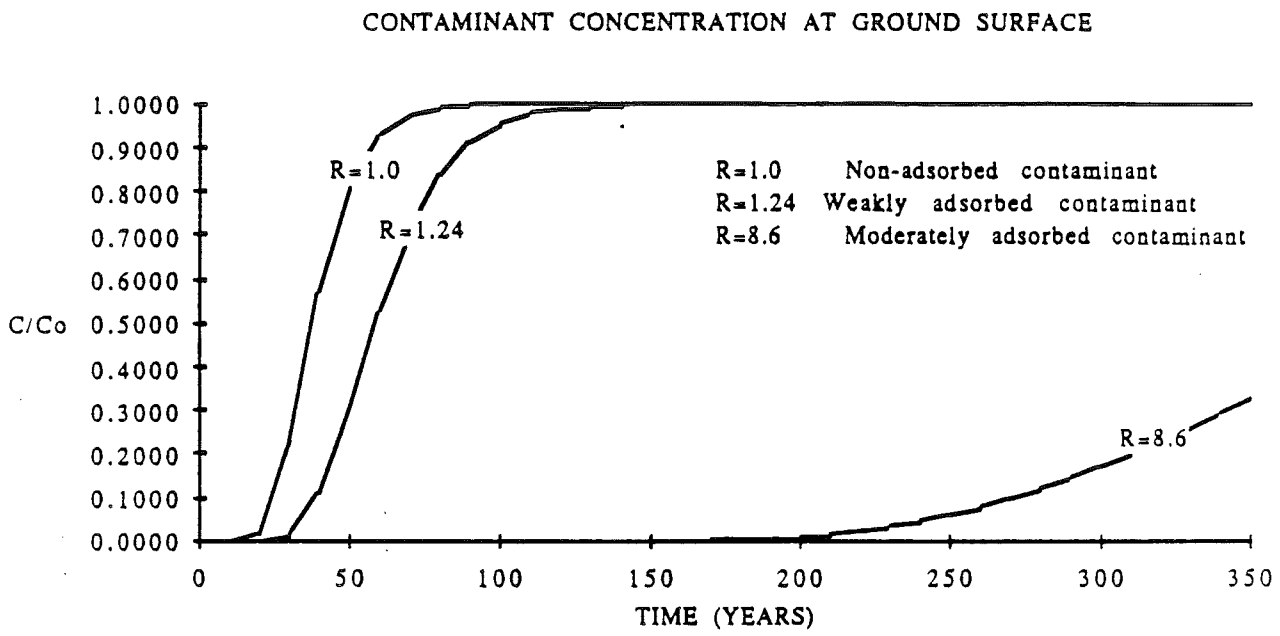
FIGURE 15

UPWARD MIGRATION THROUGH THE DISCHARGE ZONE TILL

Uncoupled Transient Model - Constant Contaminant Input

Input Parameters:

Upward Groundwater Velocity - 9.6×10^{-7} cm/s
Thickness of Discharge Zone Till - 12 m
Dispersivity - 0.6 m
Coefficient of Molecular Diffusion (R=1) - 1.5×10^{-6} cm^2/s



8.6 Coupling of Component Models

The individual component models described in the preceding sections serve to illustrate the principal features of groundwater contaminant migration at Site F. The key pathway for the off-site migration of contaminants from the landfill through the groundwater will be downward through the Weathered Shale and Weakly Fractured Shale, laterally through the Highly Fractured Shale and upward through the Discharge Zone Till to the ground surface. This contaminant pathway is best illustrated by integrating the individual component models for the Weakly Fractured Shale and the Discharge Zone Till in series so that the overall impact of leachate from the landfill can be assessed at the ground surface at the discharge zone. Along this pathway, the contaminant transport rate will be principally controlled by the least permeable units. The contaminant transport rates in the Weathered Shale and the Highly Fractured Shale are extremely rapid compared to the Weakly Fractured Shale and the Discharge Zone Till and will have a negligible effect on the contaminant concentrations existing at the discharge zone. For simplicity, only the Weakly Fractured Shale and the Discharge Zone Till component models were integrated and transport through the Weathered Shale and Highly Fractured Shale was assumed to be instantaneous.

The individual component models as presented assume a constant contaminant input concentration which does not vary

with time. However, the concentration input to the Highly Fractured Shale from the overlying Weakly Fractured Shale will vary with time as contaminants migrate from the landfill through the Weakly Fractured Shale. The time-varying concentration can be simulated in the models by the method of superposition. For these models, the time-varying input concentration is discretized into several concentration-time steps. For each concentration-time step the contaminant transport model is solved using the appropriate concentration and time interval, and then the concentration-time step solutions are superimposed (by addition or subtraction as appropriate) to provide the integrated solution. A constant input of 1500 mg/L chloride was assumed to be derived from the landfill. For conservatism, this concentration is greater than the chloride level presently observed in the leachate at the Burlington Landfill (1200 mg/L). The time-varying chloride concentration exiting the Weakly Fractured Shale was discretized and input into the Discharge Zone Till for calculation of the chloride concentrations which would occur at the ground surface in the discharge zone.

Figure 16 shows the time-varying concentration at the ground surface in the discharge zone for a non-absorbed contaminant such as chloride. As shown in Figures 13a and 15, the time-varying concentration for a weakly adsorbed organic contaminant such as methyl ethyl ketone or methylene

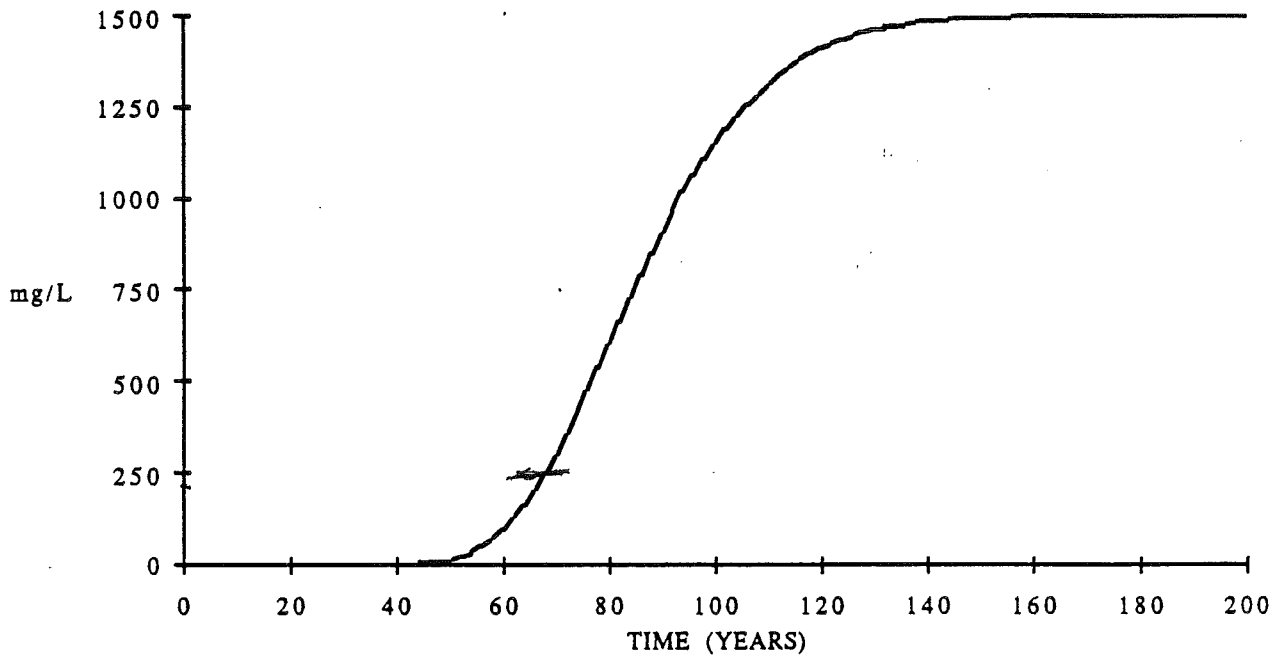
FIGURE 16

TIME-VARYING CONCENTRATION AT THE GROUNDSURFACE IN THE DISCHARGE ZONE

Coupled Transient Model - Constant Chloride Input 1500 mg/L

Incorporates Weakly Fractured Shale Model Figure 13a and Discharge Zone Till Model Figure 15. Output of the Weakly Fractured Shale Model forms input to the Discharge Zone Till Model.

CHLORIDE CONCENTRATION IN GROUNDWATER DISCHARGING TO THE GROUND SURFACE AT THE DISCHARGE ZONE



chloride would be expected to be similar. The time of first arrival of contaminants (C/CO=0.1 or chloride=150 mg/L) at the Discharge Zone Till occurs at approximately 60 years. The arrival time estimated from the integrated model is similar to the sum of the first arrivals for the individual component models. Chloride concentrations at the discharge zone till reach the input concentrations at approximately 130 years. Arrivals times for moderately adsorbed contaminants such as xylene and trichloroethylene may be longer by a factor of 15. The input concentration will always be eventually achieved for conservative contaminants (i.e. those which are not affected by degradation or transformation reactions) with a constant input from the landfill.

Based on observations at landfill sites and in laboratory leaching experiments, the assumption that contaminant inputs from landfills will remain constant for periods of 200 years or more is inappropriate. It is generally regarded that contaminant concentrations in municipal landfill leachate will be initially high during operation and shortly following closure, but will decline as more readily leachable compounds are removed. Within the landfill, many organic compounds will undergo biological and chemical degradation or may be volatilized and vented with the landfill gases. However, the time scale over which this decline may occur and the rate of the decline cannot be

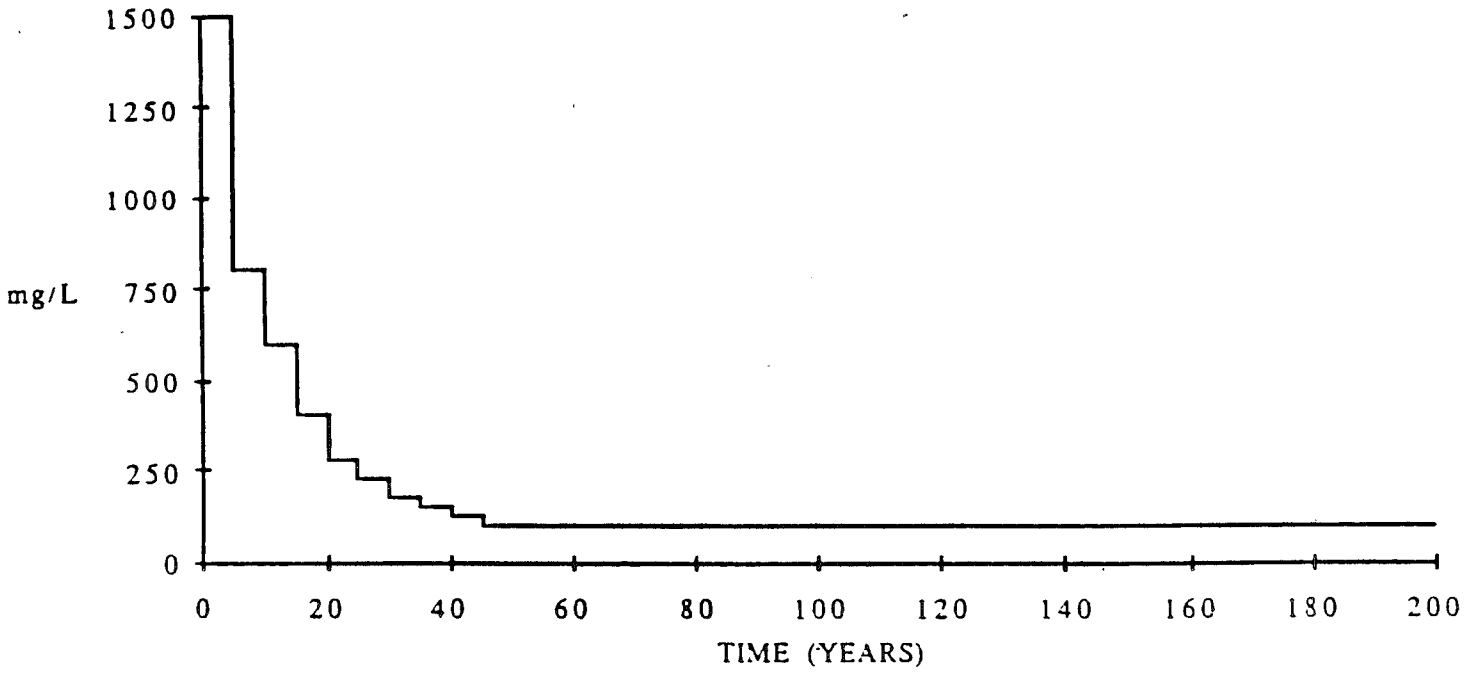
predicted with absolute confidence. Nonetheless, it is expected that declines in contaminant concentrations in landfill leachate will occur over a period of tens of years.

The effect of a declining contaminant input concentration is significant with respect to off-site migration of contaminants through the groundwater. At Site F, contaminant migration through the Weakly Fractured Shale and Discharge Zone Till occur on a scale of tens of years, similar to that of the expected declines in input concentration. Because of this relatively long travel time, the high initial contaminant concentrations in the leachate may never be reflected at the discharge zone, even for non-absorbed contaminants. This effect can be illustrated by examining the integrated model with a decreasing contaminant input. Figure 17 shows an example of decreasing chloride concentration with time which was used to represent the input from the landfill in the following simulations. Chloride is initially at 1500 mg/L and decreases to 100 mg/L after 50 years.

At the present time, there is no field or laboratory evidence to suggest the same pattern of declining concentrations for organic chemical contaminants derived from municipal landfills. Some declines would be expected, however, and for the purpose of these simulations similar patterns of decline were assumed.

FIGURE 17

CHLORIDE CONCENTRATIONS IN LANDFILL LEACHATE



For the following simulations, initial input concentrations of 1000 ug/L and 200 ug/L were assumed for weakly adsorbed organic contaminants such as methyl ethyl ketone and methylene chloride. Although concentrations of these contaminants in the range of 1000 ug/L are observed at the present Regional Landfill in Burlington, they cannot be considered typical in municipal landfill leachates. At the Burlington Landfill such high concentrations must reflect the past disposal of industrial wastes in the landfill. In the management of future municipal landfills, it is anticipated that such disposal will be restricted and the concentrations of these organic chemicals will be in the tens to several hundreds of micrograms per litre as they are at most other municipal landfill sites in Ontario.

The concentrations of chloride, weakly adsorbed organic (high initial input) and weakly adsorbed organic (moderate initial input) in the groundwater discharging to the groundsurface at the Discharge Zone are shown in Figures 18, 19 and 20, respectively. The long arrival times for contaminants to reach this location have resulted in a reduction in the peak chloride concentration to approximately 350 mg/L compared to the peak input of 1500 mg/L. Associated with this reduction in peak concentration is an extension of the period of time during which elevated chloride concentrations are observed. The duration of chloride release from the source is approximately 40 years

FIGURE 18

COUPLED MODEL-CHLORIDE CONCENTRATION AT DISCHARGE ZONE-DECLINING CHLORIDE INPUT

Coupled Transient Model

Incorporates Weakly Fractured Shale Model Figure 13a and Discharge Zone Till Model 15 with a declining concentration input Figure 17. Output of the Weakly Fractured Shale Model forms input to the Discharge Zone Till Model.

CHLORIDE CONCENTRATION IN THE GROUNDWATER DISCHARGING TO THE GROUND SURFACE AT THE DISCHARGE ZONE

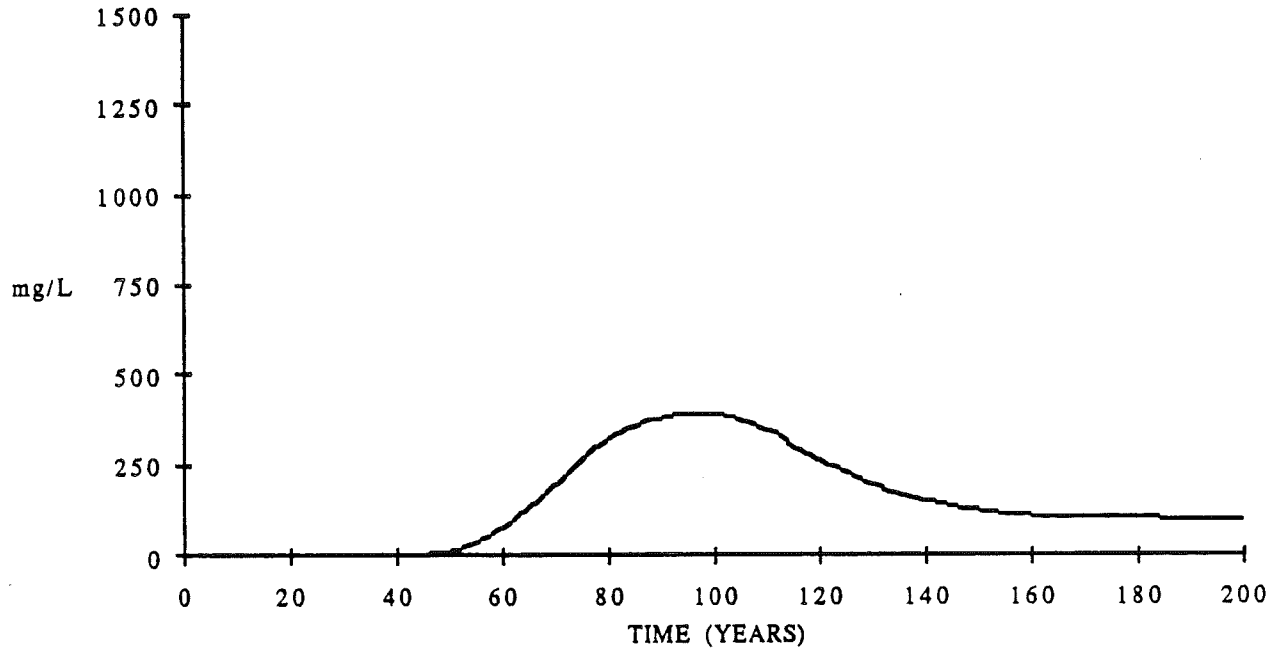


FIGURE 19

COUPLED MODEL-CONCENTRATION OF WEAKLY ADSORBED ORGANIC CONTAMINANT AT DISCHARGE ZONE-DECLINING HIGH INITIAL INPUT

Coupled Transient Model

Incorporates Weakly Fractured Shale Model Figure 13a and Discharge Zone Till Model Figure 15 with a declining concentration input of the form shown in Figure 17 with initial concentration of 1000 ug/L. Output of the Weakly Fractured Shale Model forms input to the Discharge Zone Till Model.

CONCENTRATION OF WEAKLY ADSORBED ORGANIC CONTAMINANT IN THE GROUNDWATER DISCHARGING TO THE GROUND SURFACE AT THE DISCHARGE ZONE

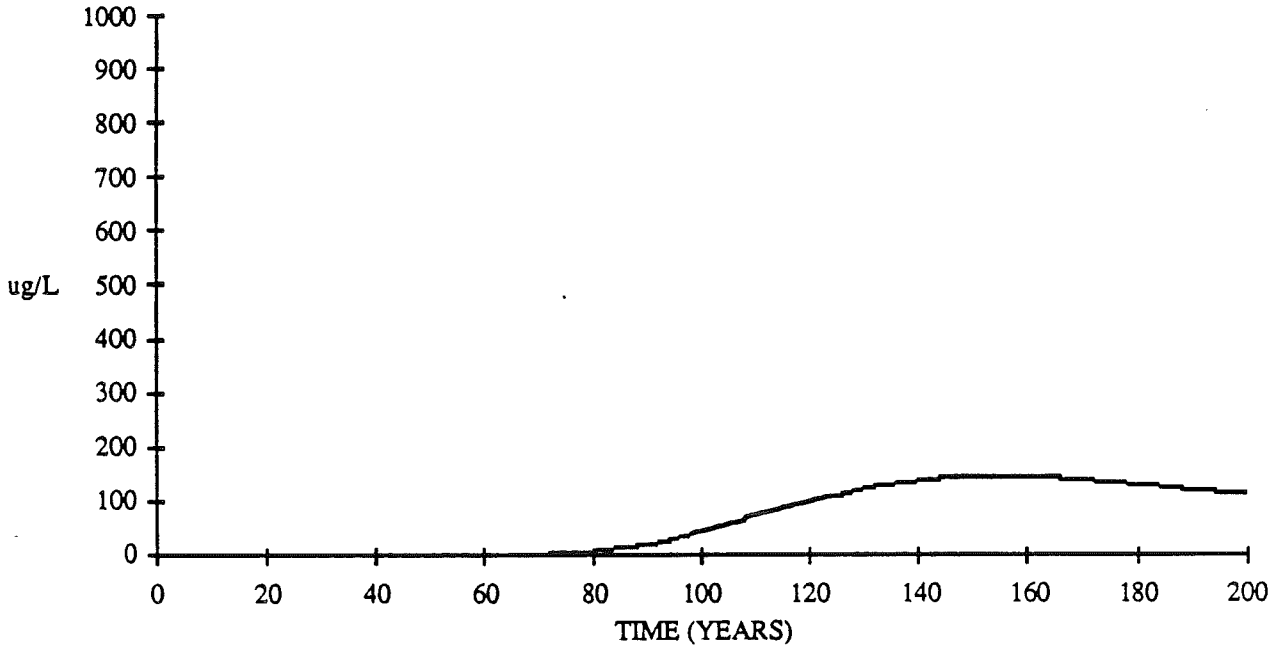


FIGURE 20

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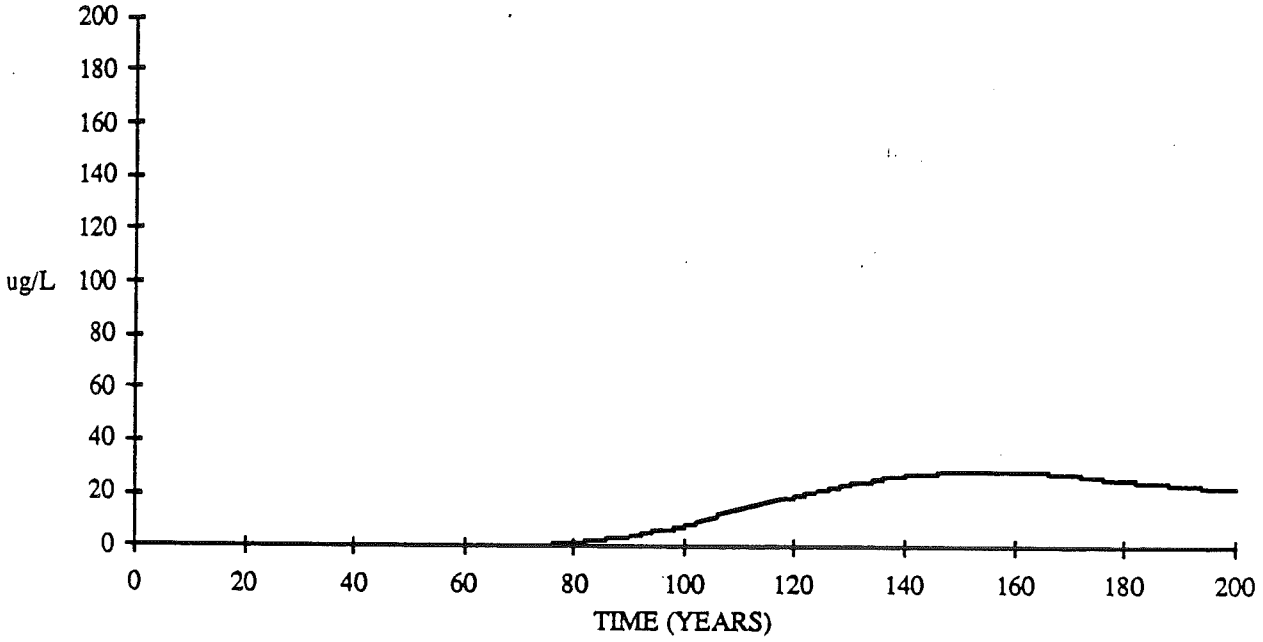
COUPLED MODEL-CONCENTRATION OF WEAKLY ADSORBED ORGANIC CONTAMINANT AT DISCHARGE ZONE-DECLINING MODERATE INITIAL INPUT

Coupled Transient Model

Incorporates Weakly Fractured Shale Model Figure 13a and Discharge Zone Till Model Figure 15 with a declining concentration input of the form in Figure 17 with initial concentration of 200 ug/L. Output of the Weakly Fractured Shale Model forms input to the Discharge Zone Till Model.

CONCENTRATION OF WEAKLY ADSORBED ORGANIC CONTAMINANT IN THE GROUNDWATER DISCHARGING TO THE GROUND SURFACE AT THE DISCHARGE ZONE



whereas the duration of elevated chloride concentrations at the site boundary is 80 to 90 years. Trends in the concentrations of weakly adsorbed organic contaminants have lower peak concentrations, longer peak durations and later peak arrivals due to the effects of dispersion and adsorption. These results indicate that there will be an impact at the Discharge Zone, but for a declining source the peak concentrations will be subdued.

9.0 LANDFILL DESIGN CONSIDERATIONS

9.1 Hydrogeologic Factors

A landfill constructed at Site F would be predominantly founded in the Queenston shale.

Contaminant pathway analysis and interpretation indicates that there is essentially one pathway by which leachate might migrate off-site with groundwater. The pathway consists of the following components:

a) Vertical migration downward through the weathered shale and the weakly fractured massive shale.

b) Lateral migration through the highly fractured shale.

c) Upward migration through the Discharge Zone Till.

This potential contaminant pathway is illustrated in Figure 11.

Leachate migrating laterally through the weathered shale would discharge on-site and would be collected for treatment.

The dominant hydrostratigraphic unit controlling leachate migration at Site F is the weakly fractured massive shale. Tests show that this unit has relatively good attributes for waste disposal. It likely has good attenuation abilities and a low hydraulic conductivity (no more than 1×10^{-7} cm/s). Farther along the flow system, the Discharge Zone

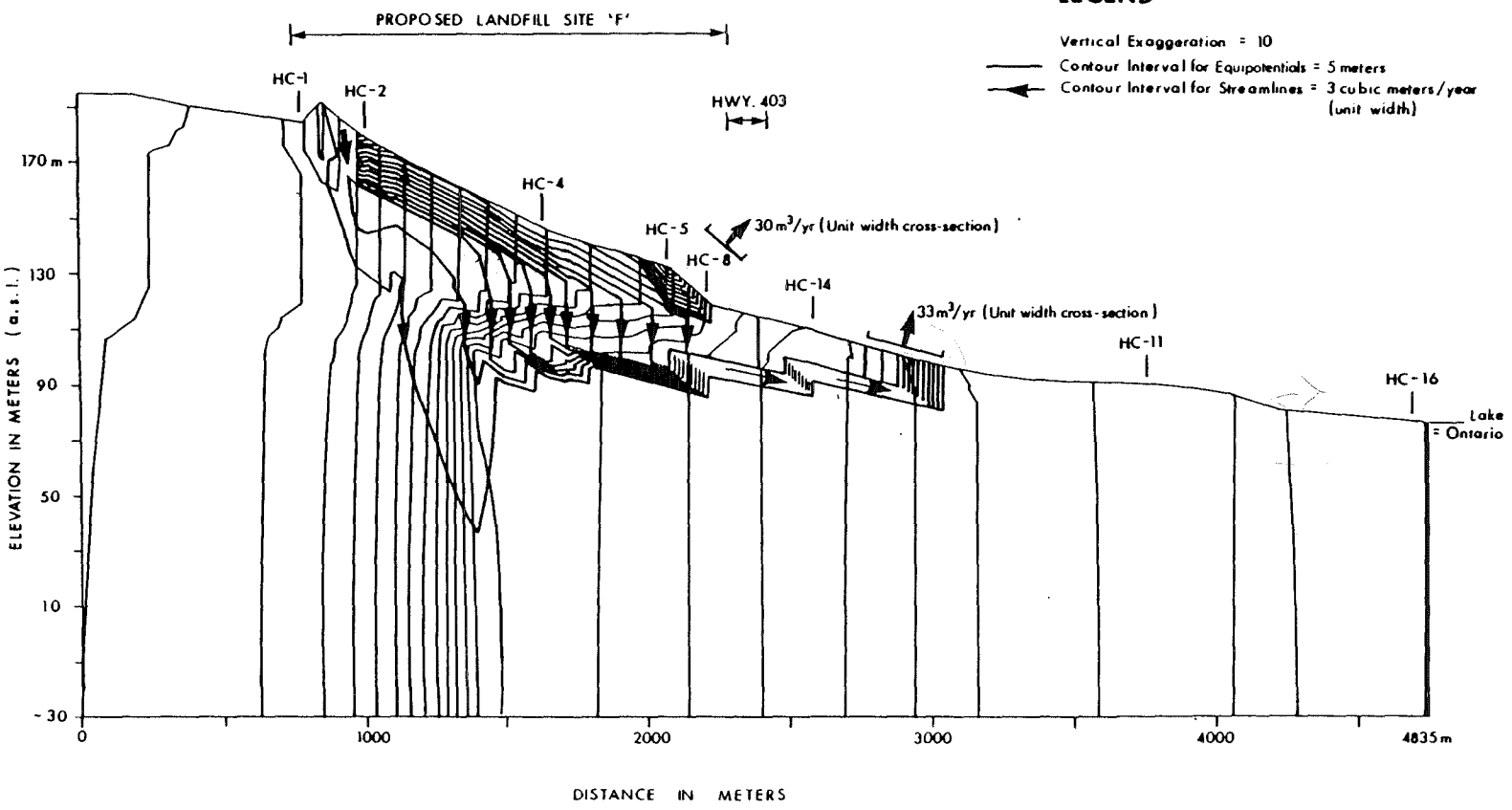
Till will provide a considerable additional capability for the natural attenuation of leachate.

In order to evaluate potential environmental impacts, approximate hydraulic characteristics of a landfill were superimposed on the calibrated computer simulation of the natural groundwater flow system. For the simulation, the water table elevation was increased uniformly by an additional 10 meters across the landfill site, and a hydraulic conductivity of 10^{-4} cm/s was assigned to the layer which represented the landfill refuse. All other variables from the calibrated flow model remained unchanged. The results of the simulation are presented in Figure 21.

The average infiltration into the underlying Weathered Shale beneath the fill increased to 22.6 mm/year, for a net increase of 8.9 mm/year over the simulation for natural conditions. The discharge from the Highly Fractured Shale Unit south of Highway 403 was about 17,000 m^3/year, for a net increase of 3,000 m^3/year (1.3 gpm) above the simulated natural discharge in that area. In addition, the model indicates that there would be a leachate discharge on the order of 15,000 m^3/year (6.3 gpm) at the downgradient toe of the landfill.

FIGURE 21

LANDFILL SUPERIMPOSED ON COMPUTER SIMULATION OF GROUNDWATER FLOW SYSTEM



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9.2 Landfill Operation

The method of landfill operation proposed for the site is a combination of the trench and area fill methods. Deep extensive excavations will be dug into the Weathered Shale to provide a portion of the required landfill capacity and to provide material for cover and earth works. The trench will be progressively filled with waste and an area fill will be constructed on top of the trench to an average height of about 15 m above existing grade. The waste will be spread in thin layers, compacted and covered with earth daily.

Modern sanitary landfills are constructed using engineered methods to protect the environment. When rain water or groundwater infiltrate garbage a mineralized leachate is formed that can adversely impact groundwater and surface water if not adequately controlled by natural or engineered methods. Further, landfill gases are formed which, under certain circumstances, can be explosive or odorous. These too must be controlled by natural or mechanical means in a properly designed and engineered landfill.

9.3 Leachate Control

Leachate control systems vary in complexity depending upon site specific hydrogeologic conditions. Given the hydrogeologic condition of Site F, two engineered methods (primary systems) to protect the environment were considered. These were a leachate underdrain system and a liner. Liner considerations are discussed in Section 9.3.1.

At large above ground landfills, leachate underdrain collection systems are usually installed as a precautionary measure even if impact estimates indicate that less efficient collection systems such as toe or perimeter drains backed up by natural attenuation processes would suffice.

At Site F, a leachate underdrain system will be installed to collect the leachate in the landfill and at the toe of the landfill. In order to minimize leachate mounding in the refuse, a 50 m tile spacing was adopted. The leachate collection design necessary for this purpose is described in the Design and Operations Report.

With the disturbance of natural soils and changes to surface grades brought about by landfilling, it is anticipated that the annual infiltration - leachate production rate through the final landfill cover could be as much as 305 mm per annum. Most of the leachate will be collected by the underdrains and will be treated in an appropriate manner. The underdrains will not significantly affect the prevailing recharge gradients at the site and consequently some leachate will move through the Weakly Fractured Massive Shale Unit into the Highly Fractured Shale Unit. The quantity of leachate egress predicted by the model would be in the 13.7 mm to 22.6 mm/year range. The resultant capture efficiency of the underdrain system would therefore be in the 93 to 96 percent range.

9.3.1 Liner Considerations

General

Liners are considered for the protection of water resources in hydrogeologic environments where significant off-site contamination can occur (e.g., high hydraulic conductivity and low attenuation capability) or where there are significant hydrogeologic complexities that require special treatment. While interpretations indicate that Site F does not fall into these categories, Hydrology Consultants was requested by the Halton staff to consider the need for liner installation to augment the system of leachate collection and natural attenuation proposed for the site design.

In general, two types of materials can be considered for liners, synthetic membranes and natural clays. Synthetic membranes have not been installed in landfills in Ontario. They have been installed in hazardous waste disposal sites in the U.S., but to-date, the experience with their integrity has not been verified.

The main advantage of a synthetic liner is its very low hydraulic conductivity (10^{-11} cm/s). The main disadvantage is the difficulty of installing a non-leaky membrane and some uncertainty about longevity of materials.

Natural clay liners have been installed at landfills in Ontario. The materials for their construction usually

comprise clayey sediments such as glacial till, that are obtained locally. In one case, an imported swelling clay (bentonite) was used.

Currently, clay liners are favoured in Ontario, where they are required. While these materials have a much higher hydraulic conductivity than synthetics (10^{-7} to 10^{-8} cm/s), their much greater thickness (1.0 - 1.5 m) results in relatively low flows because of low hydraulic gradients. As indicated by Milligan (1983), a 1 mm thick synthetic liner with a permeability of 7.5×10^{-11} cm/s and 2 m of head will pass about as much leachate as a 1.0 m thick clay liner with a hydraulic conductivity of 5×10^{-8} cm/s at the same head.

An obvious additional advantage of a clay liner is that it affords some natural attenuation capability.

Given the state-of-the-art in liner construction and the reliability of the materials, a natural clay liner would be the favoured alternative for Site F, if indeed a liner was needed.

Availability of Material

The construction of a clay liner at Site F could involve:

a) the installation, compaction and testing of a 1.0 m thick glacial till liner;

b) the installation of a 0.5 m thick sand layer on top of the liner to protect it from desiccation by moisture loss;

c) the installation of sophisticated monitoring devices to measure the effectiveness of the liner; and

d) the construction of a leachate collection system on top of the liner.

There is no sand on site for the desiccation control blanket. These materials would have to be imported. Data indicate that there is sufficient clayey glacial till at the north end of Site F to construct a 1 m thick liner over the entire disposal area. The clay content of the glacial till varies from 18 to 45 percent. At Metro Toronto's Keele Valley Landfill, experience has shown that a similar glacial till containing 20 percent clay can be compacted to achieve a hydraulic conductivity of about 10^{-8} cm/s. Therefore, it is reasonable to assume that the on-site till can be used to produce a liner with a hydraulic conductivity in the 10^{-7} to 10^{-8} cm/s range.

In accordance with the Ministry of the Environment policy, a contingency system for the liner would be required. This would comprise the natural attenuation afforded by the underlying Weakly Fractured Massive Shale augmented by a

purge well system if needed, i.e., the same contingency system as proposed for the unlined site design.

Liner Performance Estimates

The hydrogeologic interpretation and modelling indicate that most of the precipitation that falls on Site F runs off or is discharged through the shallower flow systems to the gullies. The shallow groundwater component of flow would be collected by a leachate sub drain tile collector system. However, the tiles would not overcome the vertical recharge hydraulic gradients at the site and leakage to the deeper flow systems would continue. It would be the purpose of any liner to significantly reduce the amount of exfiltration from the landfill to the underlying zones. It should be kept in mind, however, that the natural exfiltration from the site is very low at about 10,000 m^3/year (19 L/m or 4.2 gpm).

The rate of exfiltration to the intermediate depth flow system in the Highly Fractured Shale Unit is controlled by the overlying Weakly Fractured Massive Shale Unit which has a hydraulic conductivity of about 10^{-7} cm/s. The installation of a liner with a hydraulic conductivity of about 10^{-7} cm/s would be expected to have little effect on the rate of exfiltration from the site. An approximate similation of a 10^{-7} cm/s liner was made with the model. As expected, a liner with this hydraulic conductivity has only

a minimal effect on the exfiltration rate. The resultant rate was 9,000 m^3/year (17 L/m or 3.8 gpm).

A 10^{-8} cm/s liner was also modelled and yielded an exfiltration rate in the 2500 to 4400 m^3/year range (4.8 to 8.4 L/m or 1.1 to 1.9 gpm). The difference in the capture efficiency between the 10^{-7} and 10^{-8} cm/s liners is only about 50 percent because the 10^{-7} cm/s confining layer acts to underdrain the 10^{-8} cm/s liner. This increases the hydraulic gradient through the liner and offsets the reduced hydraulic conductivity (Q = kiA).

Conclusion on Liners

Based on the following factors:

a) that the natural rate of leakage from the site to the intermediate flow system is low at about 10,000 m^3/year;

2 million gallons a year w/o liner

b) that a 10^{-8} cm/s natural clay liner would only reduce the exfiltration rate to about 4400 m^3/year;

approx 1 million w/liner

c) that the exfiltration rates in either case are relatively small;

Trust if wrong about @ + it's 2 million gallons w/o liner

d) that liners are expensive to install and require a high level of control during construction and long-term careful monitoring after construction; and

e) that the same environmental contingencies will be required in either case, comprising a relatively inexpensive, high efficiency, purge well system;

it is concluded that a liner would not be an effective method of providing additional environmental integrity to the site. Therefore, consideration of a liner for this site is not warranted or recommended.

9.4 Groundwater Quality Impact at Discharge Zone Till

Advective flow calculations show the time of first arrival of conservative contaminants including weakly-absorbed organic compounds (C/Co=0.1 or chloride = 150 mg/L) at the Discharge Zone occurs at approximately 60 to 100 years. Based on the estimated decline in the chloride concentration in the leachate, the maximum increase in chloride at the Discharge Zone would be about 350 mg/L. This is not a severe impact, and based on the following considerations, would likely never actually occur or be measured.

1) The groundwater discharge rate at the Discharge Zone Till is calculated to be relatively low and therefore contaminant loading will also be low.

2) Groundwater in the Highly Fractured Shale Unit has naturally high concentrations of inorganic constituents such as chloride (i.e., in the order of 10,000 mg/L) which make the groundwater unsuitable for drinking water supply. Consequently, contamination of the groundwater in itself does not constitute an unacceptable environmental impact with respect to Public Health and Safety.

3) The discharge would be mixed and diluted with surface water.

4) Weakly-absorbed organic volatiles in the discharge would be rapidly lost to the atmosphere on contact.

5) Weakly-absorbed organic concentrations do not bioaccumulate.

6) The contaminant model underestimates natural attenuation. Matrix diffusion and adsorption in the Weathered Shale and Highly Fractured Shale Unit were not considered.

7) The hydrogeologic setting would perform well with respect to contaminant attenuation. This is evident from the chemical data which shows no significant contaminant concentrations off-site as a result of landfilling at the two adjacent sites.

8) The high hydraulic conductivity and lateral continuity of the Highly Fractured Shale Unit will allow implementation of effective purge well systems for control of off-site migration if necessary.

10.0 CONTINGENCIES

10.1 General

The Ministry of the Environment requires that contingency control systems be considered and conceptualized at all new sanitary landfill sites.

Site F would be equipped with a primary leachate collection system to protect the aquatic environment (groundwater and surface water). In addition, contingencies are available to serve as back-ups to the primary system, namely, natural attenuation, purge wells and perimeter drains.

10.2 Groundwater

Section 8 of this report deals at length with the attenuating capabilities of the hydrostratigraphic units at Site F and the environmental protection they afford. Monitoring would determine the effectiveness of the primary collector system (underdrains) and the primary back-up system (natural attenuation). In the event that these systems were to fail a secondary back-up system comprising purge wells and perimeter drains could be installed.

10.2.1 Natural Attenuation

As discussed in Section 9.4 the natural hydrogeologic environment would renovate contaminants to a considerable degree. There would be about an 80 percent reduction in peak contaminant concentrations as a result of natural attenuation.

10.2.2 Purge Wells

The feasibility and effectiveness of leachate control and collection by means of a purge well system has been established by the PW1 pumping test results. The radius of the cone of hydraulic influence generated from pumping at a rate of about 15 L/m (3 gpm) is conservatively estimated from the data to be 150 m. Based on this and the hydraulic coefficients determined during the pumping test, it is estimated that 3 or 4 purge wells would control contaminant migration in the Highly Fractured Shale Unit. The wells would be installed in the landfill buffer area along the North Service Road to a bottom elevation of about 118 m.a.s.l. The amount of pumpage required from the wells would be in the order of 15,000 m^3/year (7.3 gpm).

The actual design of the system would require field testing and adjustment as required.

10.3 Surface Water

10.3.1 Perimeter Leachate Tile Collector System

The computer simulation of a landfill superimposed on the groundwater flow system identifies the need for the primary leachate collection system. Without it, leachate seeps would develop at the toe of the landfills and move into surface water courses. The primary underdrain system will collect this leachate.

As a backup to the underdrain system, a perimeter tile collector could be installed along the edge of the landfill if monitoring indicated its need. The system would drain to the leachate pumping station for storage and transmission to the sewage treatment plant.

11.0 MONITORING AND REPORTING

11.1 General

The goals of the site monitoring program are to monitor water quality in the vicinity of the landfill to assess the adequacy of the landfill design and to protect local groundwater supply zones and surface water from impairment due to any unforeseen circumstances.

Data collected from groundwater monitors located close to the landfill cells and from monitoring wells located within the property boundaries will be regularly assessed for any trends indicating adverse changes in groundwater quality. Should the groundwater quality demonstrate evidence of an unacceptable level of impairment, a contingency program would be implemented so as to restrain the off-site migration of contaminants. Such a contingency can be installed within a year's time. The surface water sedimentation lagoons and storm water captured within landfill cells will be monitored routinely to ensure that the water quality is suitable for discharge to surface water courses.

Leachate quality within the landfill will be monitored in selected manholes.

Gas control is not believed to be necessary at this site because the site is remote from dwellings and it is

generally surrounded by waste or significant buffers. However, gas monitors will be installed adjacent to all on-site structures and in the north and south buffer zones. In addition, combustible gas alarms will be installed in all on-site structures.

The proposed monitoring program will cover a 40-year time span subdivided into a 20-year active landfilling phase and a 20-year post closure phase or until the Ministry of the Environment is satisfied that the site poses no long-term or short-term environmental hazards.

The following sections deal with the essence of the proposed monitoring program. Exact details would have to be worked out in concert with the Ministry of the Environment.

11.2 Groundwater Monitoring

The monitors for the site would include both on-site and off-site monitors. On-site wells would include nests of standpipes and piezometers to monitor the various units in the shale. Off-site wells would comprise selected downgradient domestic wells.

The high hydraulic conductivity and lateral continuity of the Highly Fractured Shale and relatively slow migration rates through the Discharge Zone Till will facilitate the monitoring of potential off-site contaminant migration. The

Highly Fractured Shale controls the bulk of the groundwater flow at Site F so that there is a reasonable certainty of identifying off-site migration by monitoring in this zone. The slow migration rate through the Discharge Zone Till will provide sufficient lead-time for the implementation of control actions if necessary to restrict off-site migration. Travel times through the Discharge Zone Till are approximately 25 to 40 years so that monitoring the High Fractured Shale will provide adequate warning of future discharge.

On-site monitors would be constructed as single nests near the cells and as groups of nests in the direction of groundwater flow capable of monitoring chemistry vs distance. On-site monitors could be constructed in one campaign following site preparation, or in a staged manner when needed.

Prior to sampling, the water levels in the wells would be measured. The wells would then be adequately purged to ensure a fresh representative water sample and finally the wells would be sampled.

Following well installation, samples would be collected for detailed background quality analysis. Subsequent sampling and analyses would be monitored quarterly or semi-annually only for landfill contaminant indicators such as conductivity, pH, alkalinity, chloride, sulphate, calcium, sodium,

magnesium, potassium, phenol and DOC (dissolved organic carbon). A volatile organic scan would be conducted annually.

Selected off-site private wells would be monitored annually. Some existing observation wells lie within areas recommended for landfilling and will require proper abandonment by removal of the well casings and grouting with concrete.

11.3 Surface Water Monitoring

Monitoring stations would be established on the intermittent site Creeks. Detailed background chemistry would be measured prior to any disposal taking place. Thereafter, landfill contaminant indicators will be measured on the same routine basis as established for groundwater. Similarly, the surface water sedimentation ponds will be monitored.

11.4 Leachate Monitoring

Several manholes will be selected to serve as monitoring stations for fluid levels within the landfill to ensure that the system operating in the desired manner. Leachate quality will be measured in the manholes or at a central collection point to provide data for the refinement of the impact model and to provide data on the anticipated decline in contaminant concentrations in the leachate with time.

11.5 Reporting

An annual report will be prepared summarizing all of the monitoring activities and results. The objective in groundwater quality monitoring is to evaluate the trends in the water chemistry. Several water quality parameters are analyzed for in each well to enable an assessment of general water quality trends and to identify spurious data. Annual reports will contain a graphical summary of the historical data from the site wells. Analytical results will be added to these diagrams annually and this graphical summary will be included in the annual report. The annual site monitoring report will be prepared by a consulting hydrogeologist and will include an interpretation of the data and recommendations for additional monitoring or remedial action, where necessary.

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DRAWINGS

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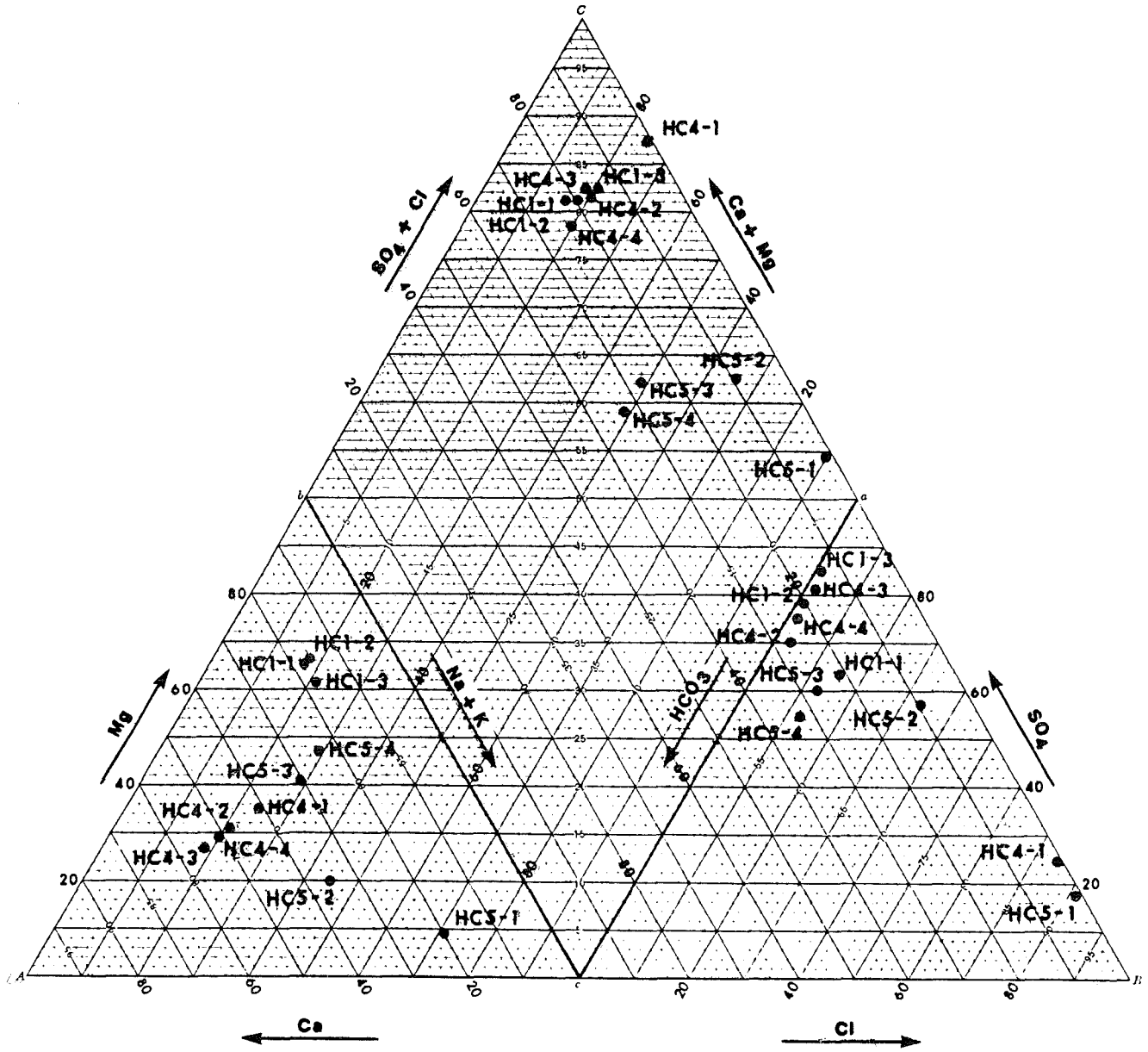
LIST OF DRAWINGS

Drawing No.	Title
1	Location Map
2	Location of Exploratory Boreholes and Water Supply Wells
3	Typical Drilling Site Well Installation
4	Natural Surface Water Drainage and Location of Surface Water Monitoring Stations
5	Location of Drilling Sites and Observation Wells
6	Regional Stratigraphic Cross-Section A-A
7	Regional Stratigraphic Cross-Section B-B
8	Local Stratigraphic Cross-Section A-a
9	Local Stratigraphic Cross-Section C-C and D-D
10	Local Stratigraphic Cross-Section E-E and F-F
11	Bedrock Topography
12	Surficial Geology
13	Regional Bedrock Potentiometric Contour Map
14	Regional Watertable Contour Map
15	Local Bedrock Potentiometric Contour Map (April, 1986)
16	Local Watertable Contour Map (April, 1986)
17	Piper Diagram
18	Water Chemistry - Anion Profile
19	Water Chemistry - Cations Profile
20	Water Chemistry - Tritium Profile
21	Geochemical Cross-Section of Chloride and Bromide
22	Monitoring System

PIPER DIAGRAM



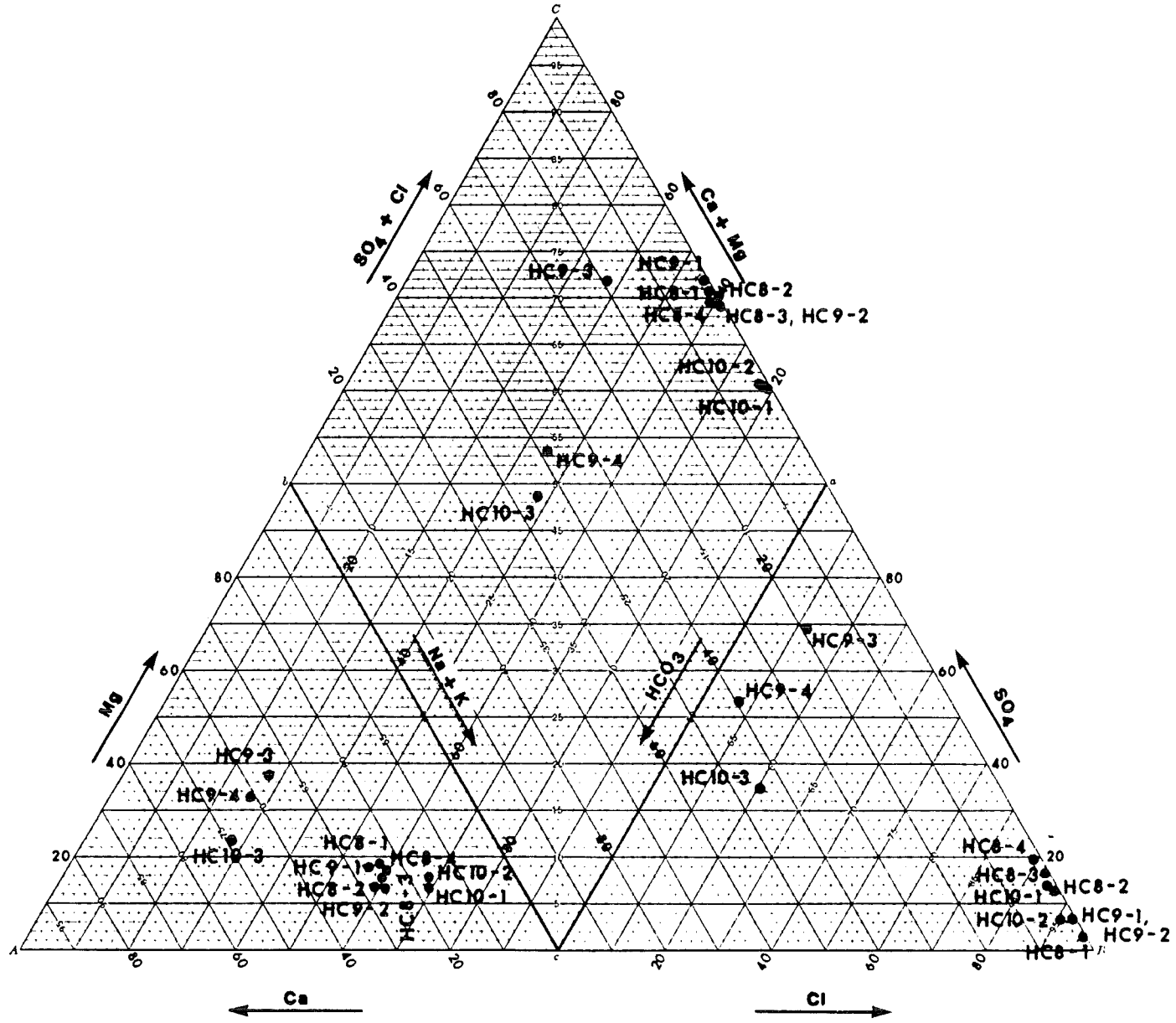
OBSERVATION WELLS	DEPTH (m)
HC1 - 1	30.5
-2	18.7
-3	5.5
HC4 - 1	30.5
-2	17.5
-3	11.4
-4	6.0
HC5 - 1	30.8
-2	18.0
-3	9.5
-4	6.2

Halton EPA - Site F

January 86 Sampling

DRAWING No 17a

PIPER DIAGRAM

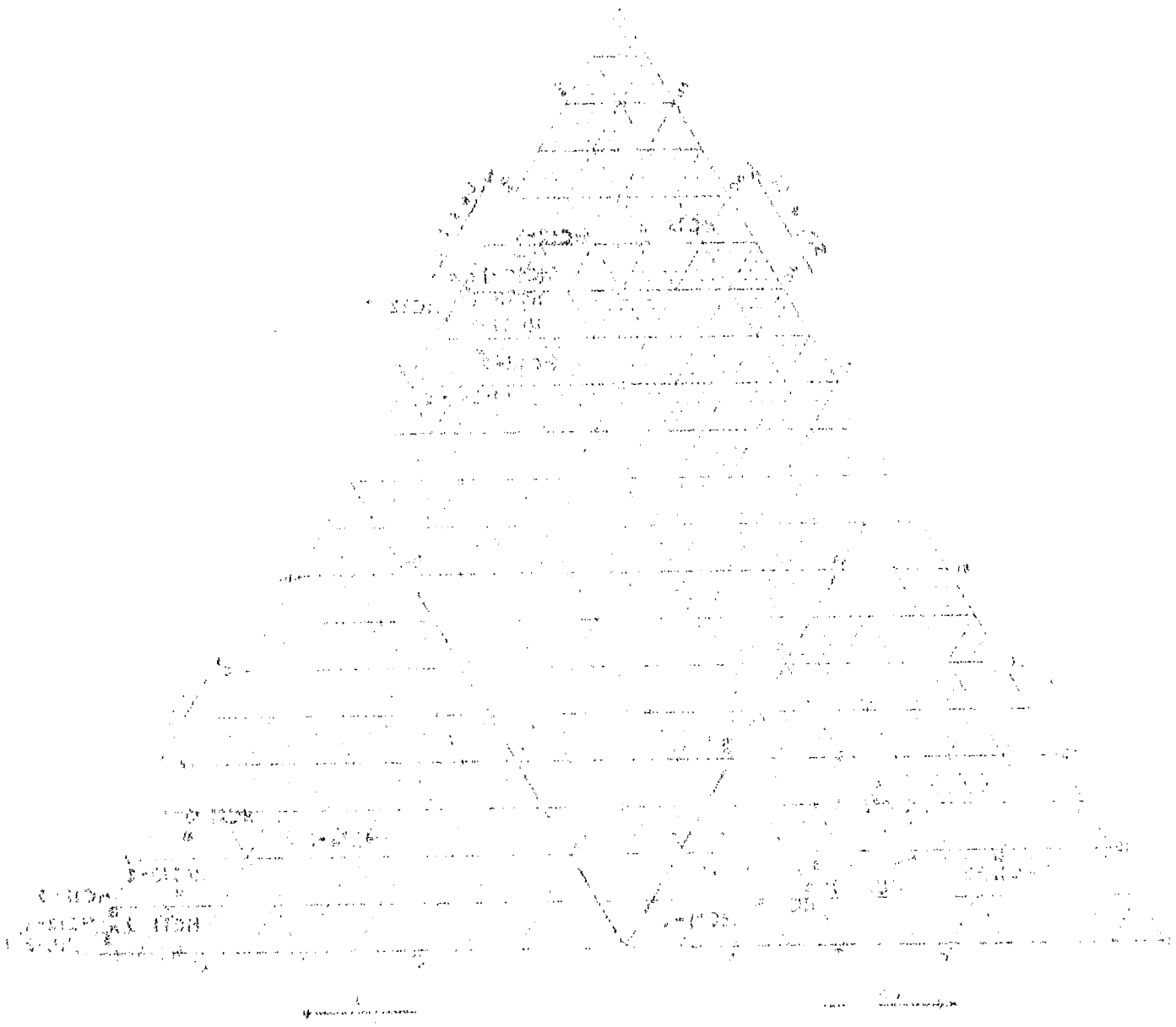


OBSERVATION WELLS	DEPTH (m)
HC8 - 1	30.9
- 2	17.2
- 3	11.1
- 4	5.6
HC9 - 1	30.4
- 2	15.9
- 3	8.9
- 4	5.1
HC10 - 1	14.4
- 2	9.5
- 3	5.1

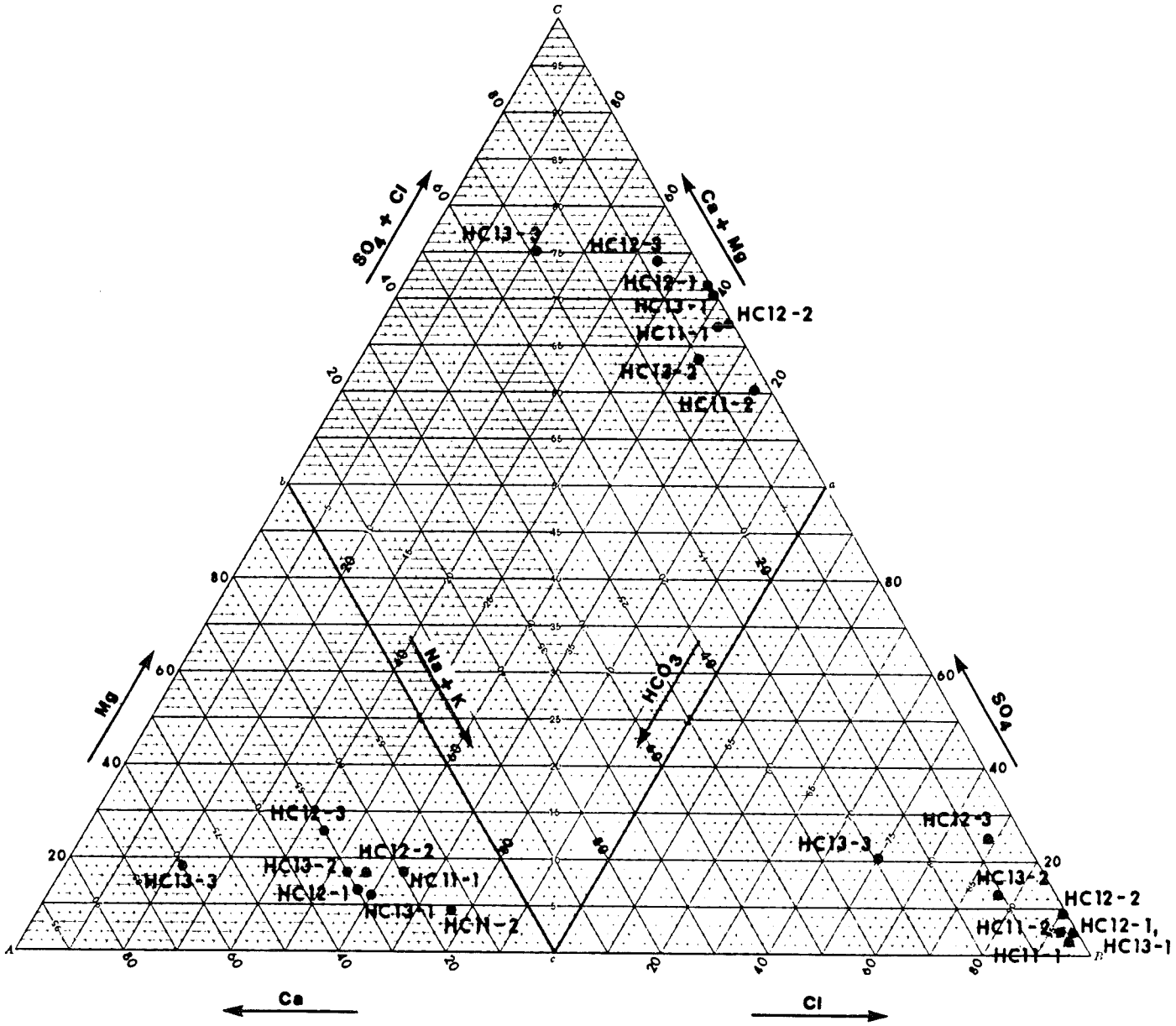
Halton EPA - Site F

January 86 Sampling

DRAWING No 17b



PIPER DIAGRAM



OBSERVATION WELLS	DEPTH (m)
HC11-1	21.6
-2	6.1
HC12-1	24.4
-2	7.8
-3	4.4
HC13-1	29.7
-2	9.2
-3	5.9

Halton EPA - Site F

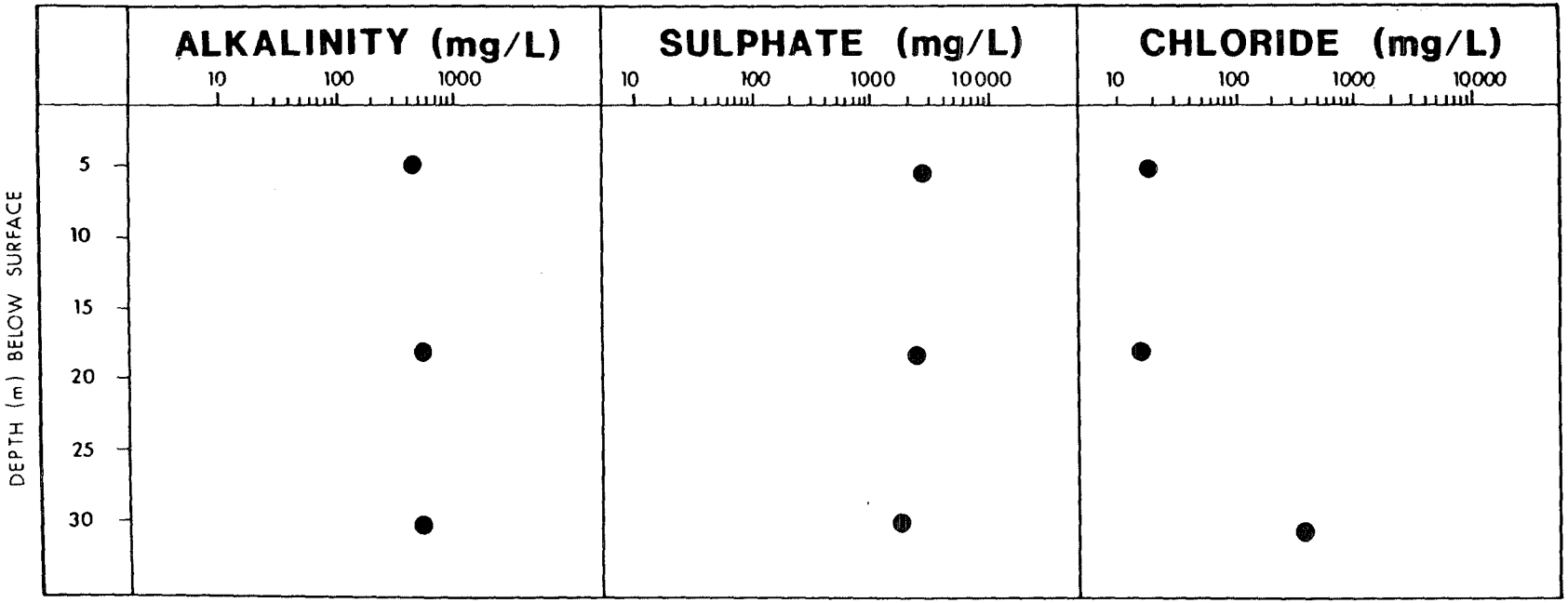
January 86 Sampling

DRAWING No **17c**

HC1-85

WATER CHEMISTRY PROFILE A - ANIONS

SITE F

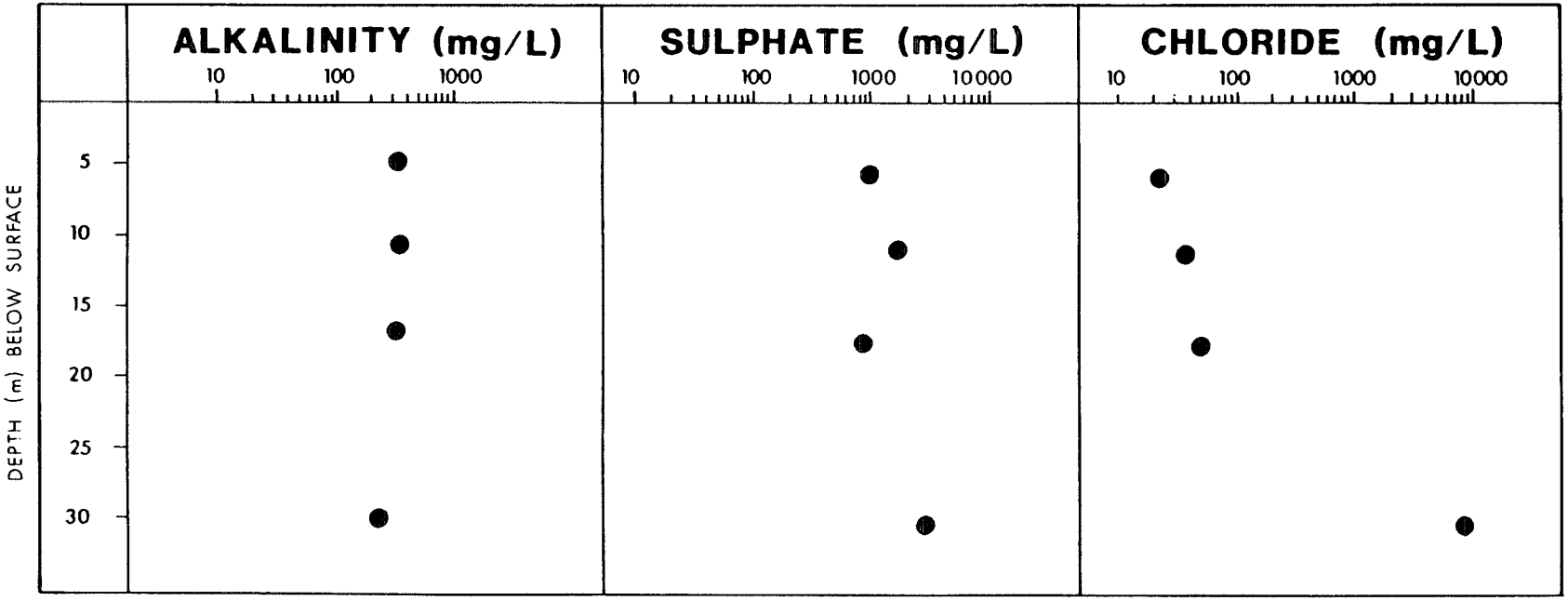


DRAWING No 18a

HC4-85

WATER CHEMISTRY PROFILE A - ANIONS

SITE F

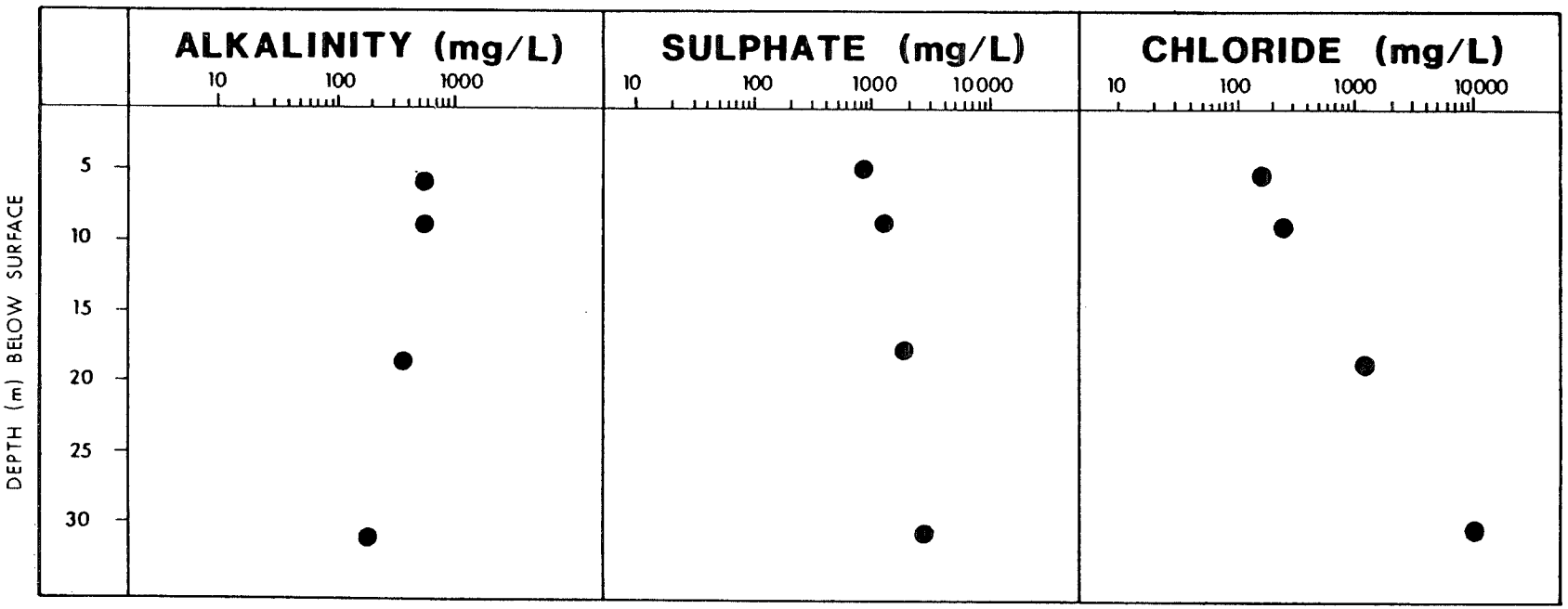


DRAWING No 18b

HC5-85

WATER CHEMISTRY PROFILE A - ANIONS

SITE F

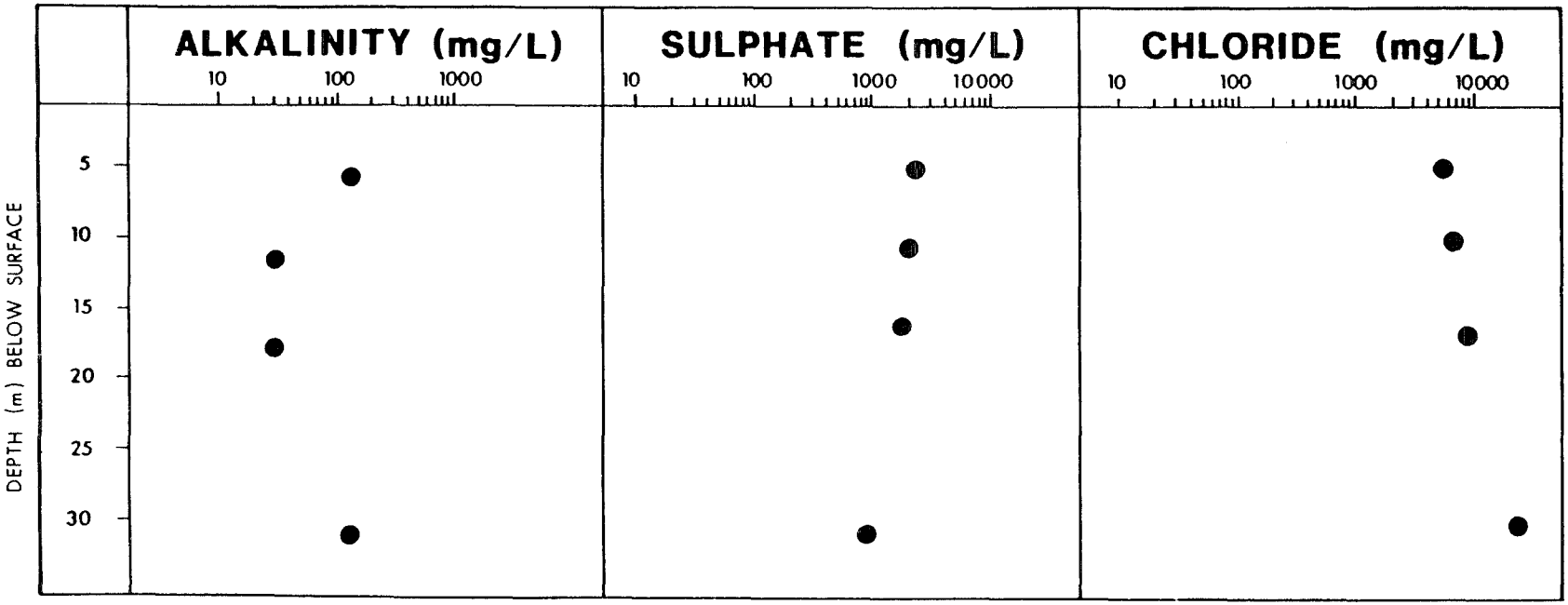


DRAWING No 18C

HC8-85

WATER CHEMISTRY PROFILE A - ANIONS

SITE F

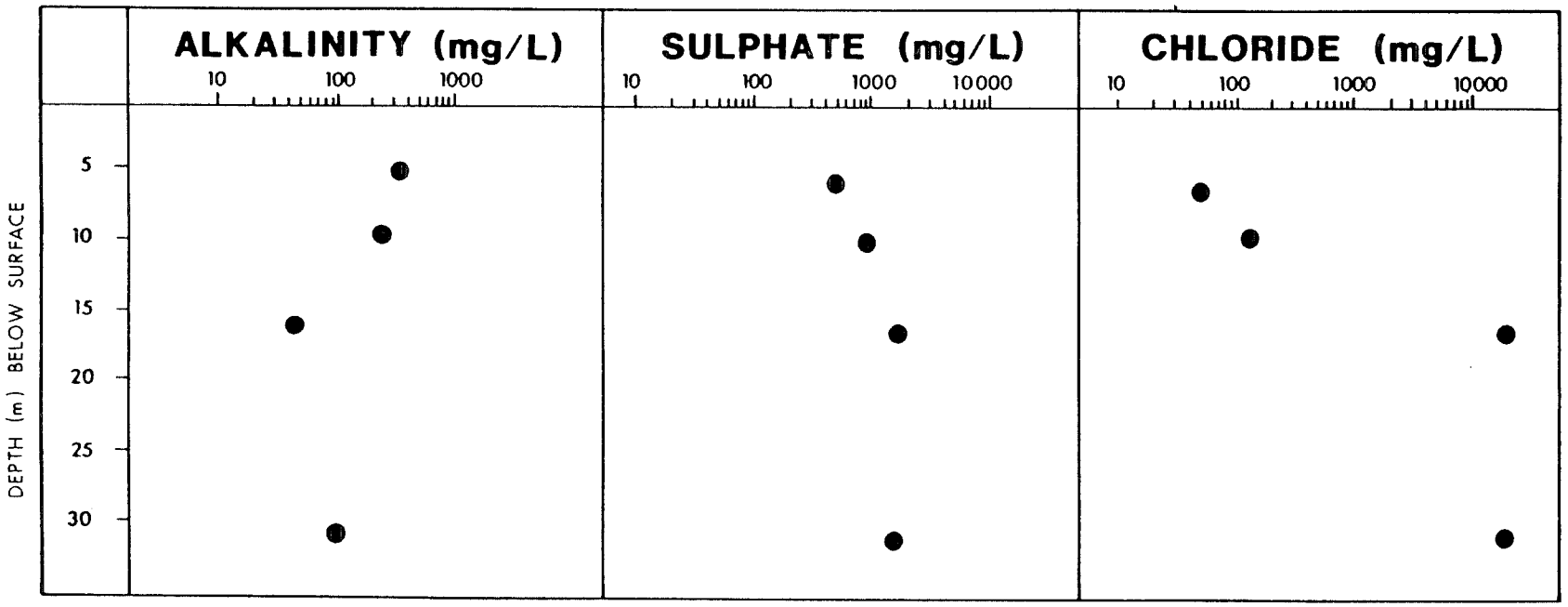


DRAWING No 18d

HC9-85

WATER CHEMISTRY PROFILE A - ANIONS

SITE F

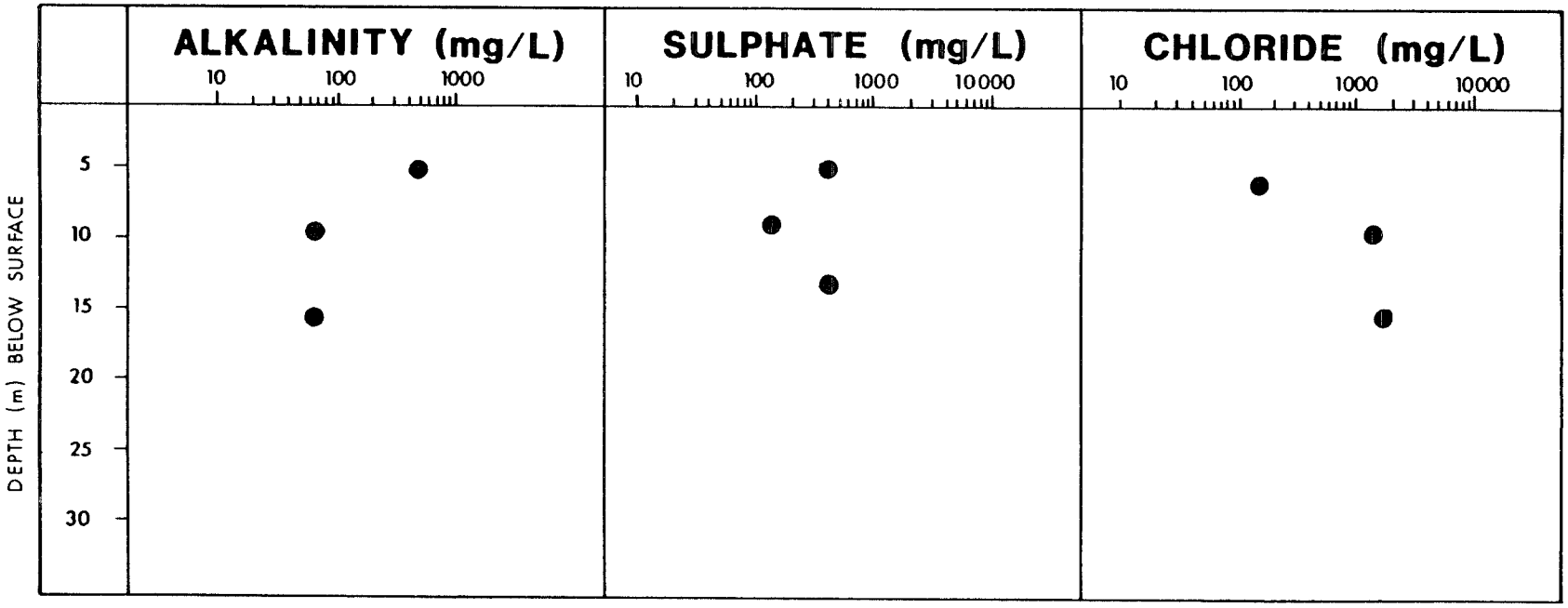


DRAWING No 18e

HC10-85

WATER CHEMISTRY PROFILE A - ANIONS

SITE F

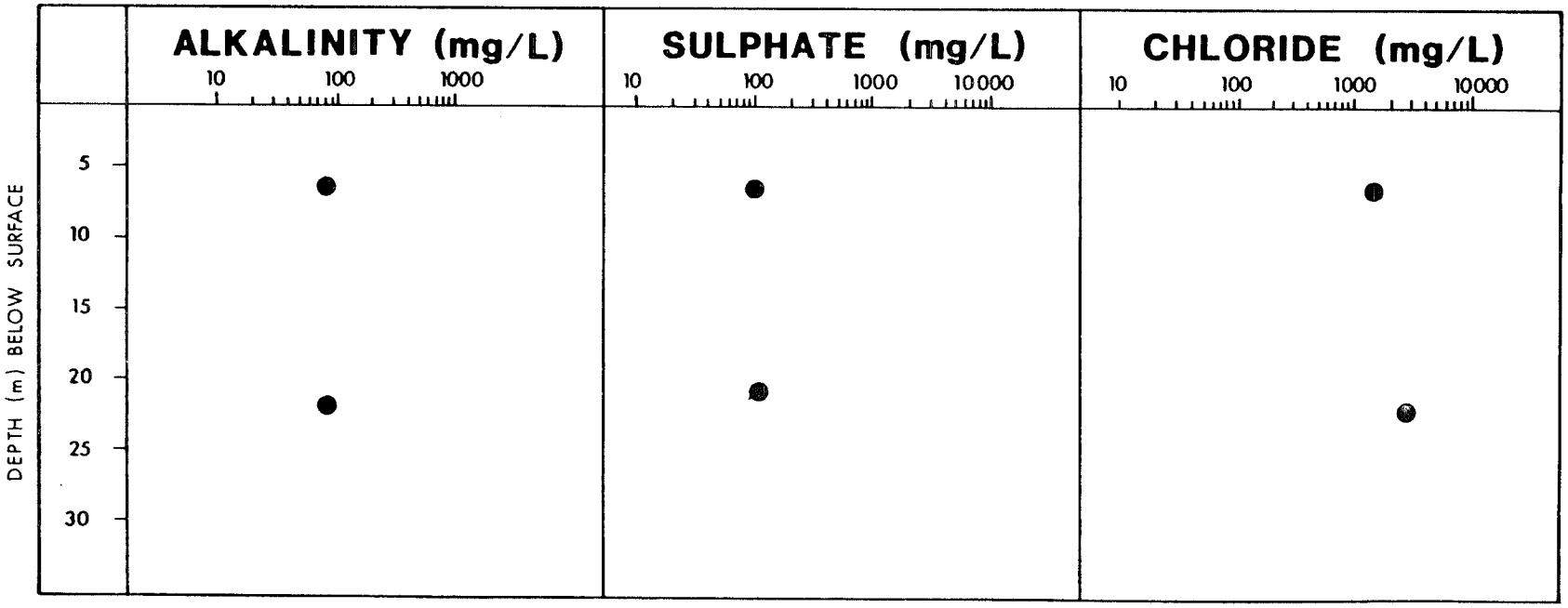


DRAWING No 18f

HC11-85

WATER CHEMISTRY PROFILE A - ANIONS

SITE F

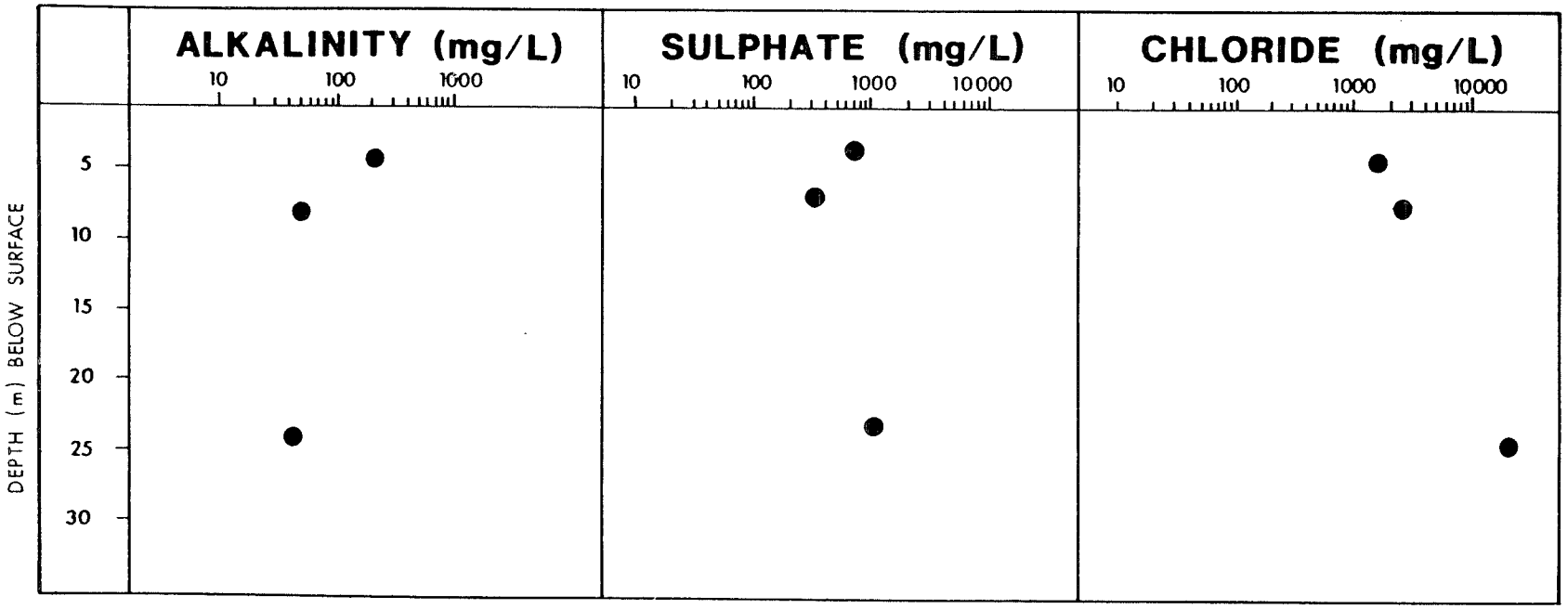


DRAWING No 18g

HC12-85

WATER CHEMISTRY PROFILE A - ANIONS

SITE F

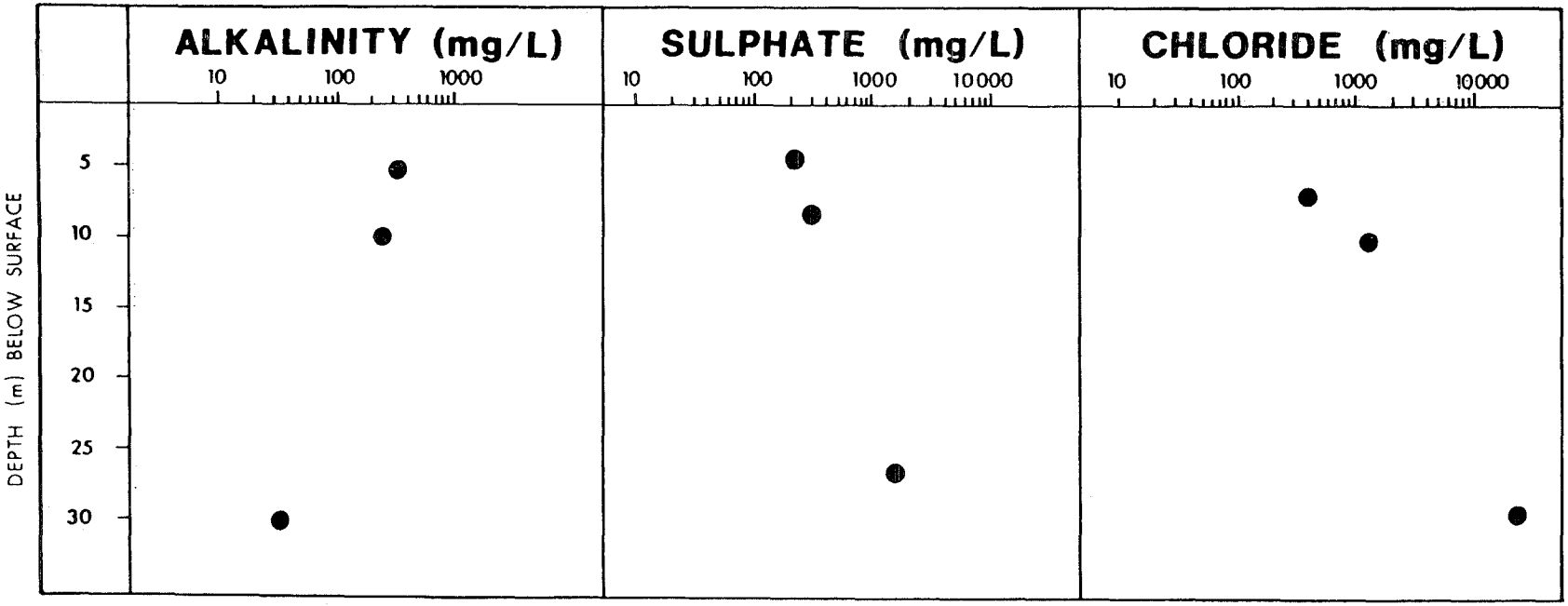


DRAWING No 18h

HC13-85

WATER CHEMISTRY PROFILE A - ANIONS

SITE F

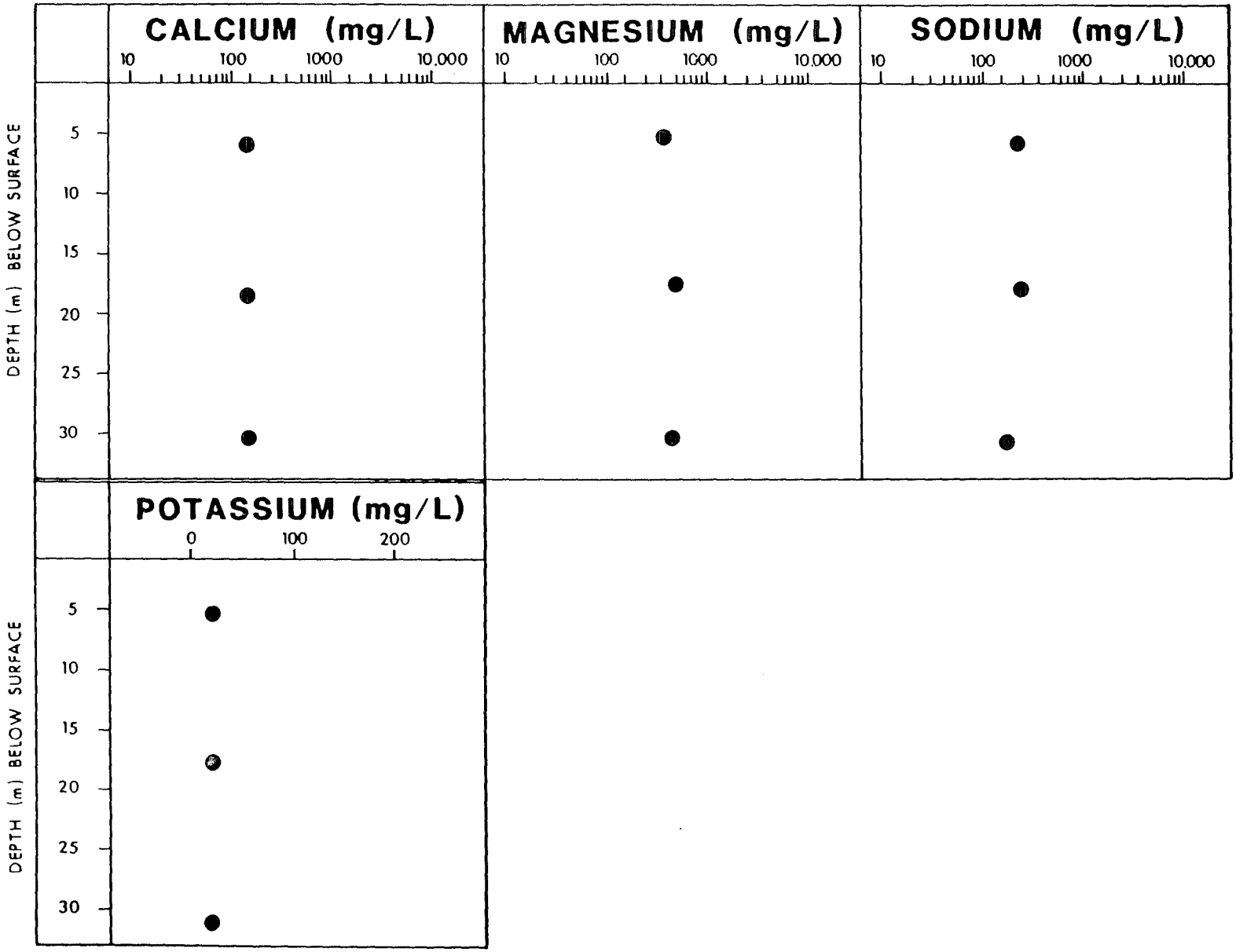


DRAWING No 18i

HC1-85

WATER CHEMISTRY PROFILE B - CATIONS

SITE F

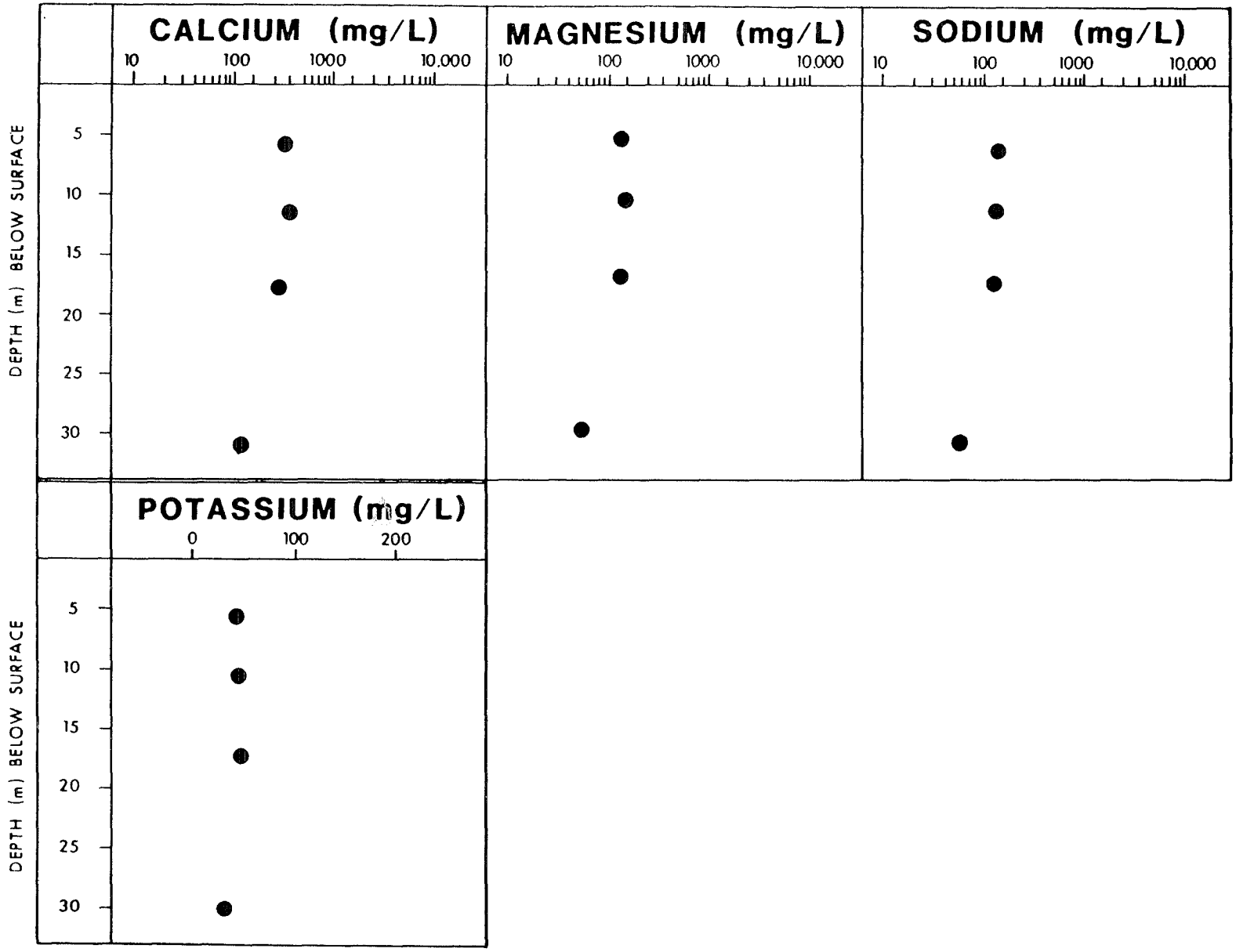


DRAWING No 19a

HC4-85

WATER CHEMISTRY PROFILE B - CATIONS

SITE F

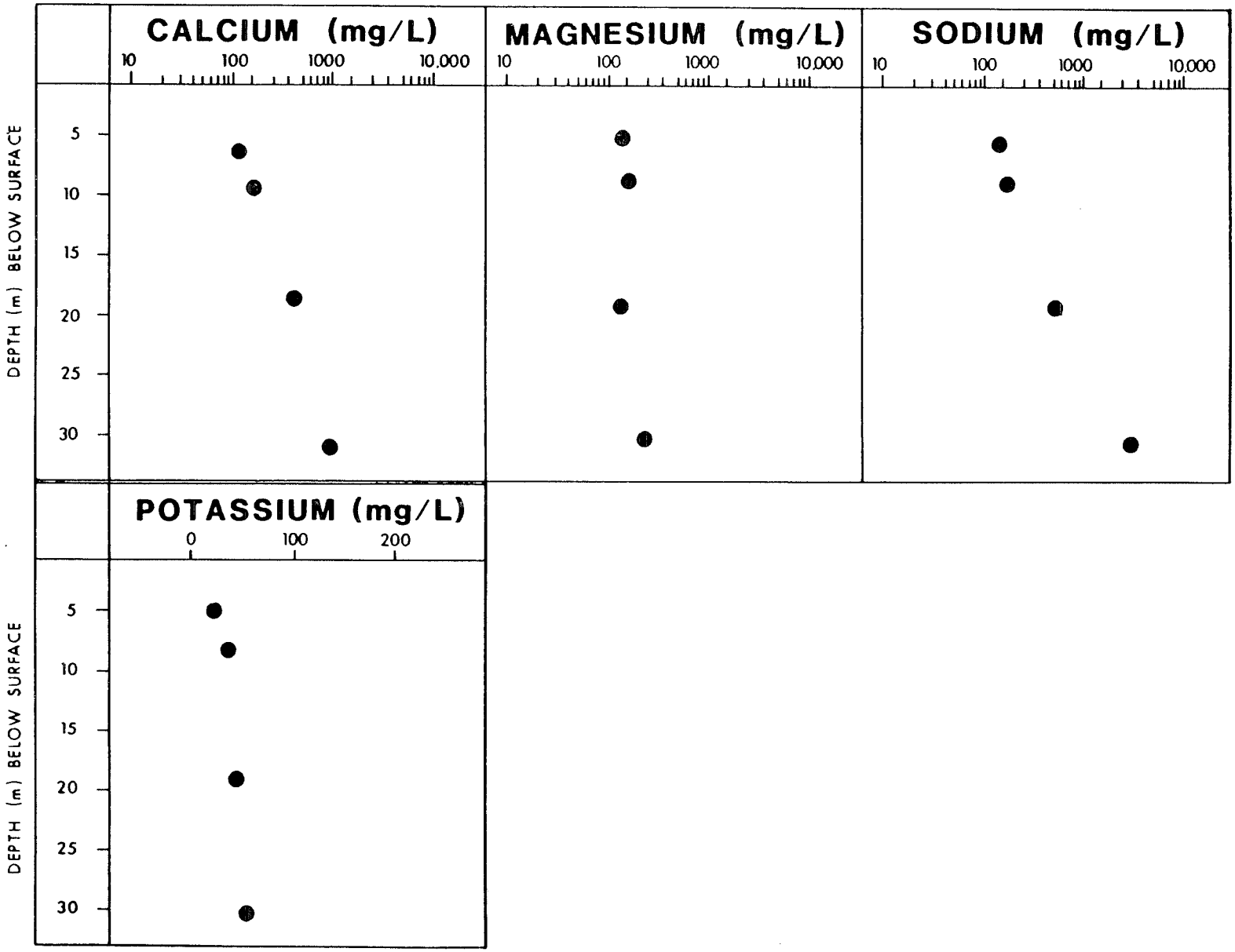


DRAWING No **19b**

HC5-85

WATER CHEMISTRY PROFILE B - CATIONS

SITE F

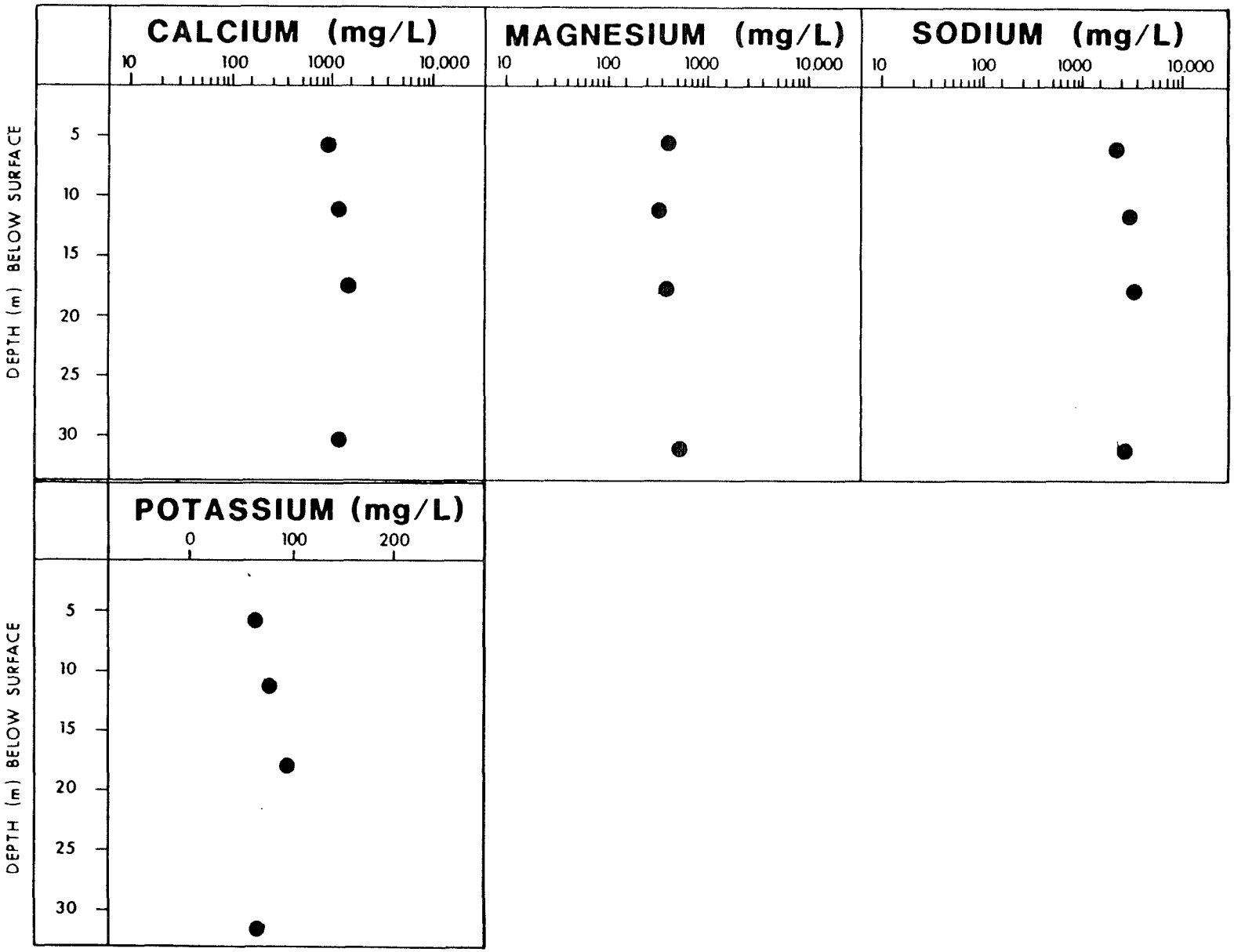


DRAWING No 19C

HC8-85

WATER CHEMISTRY PROFILE B - CATIONS

SITE F

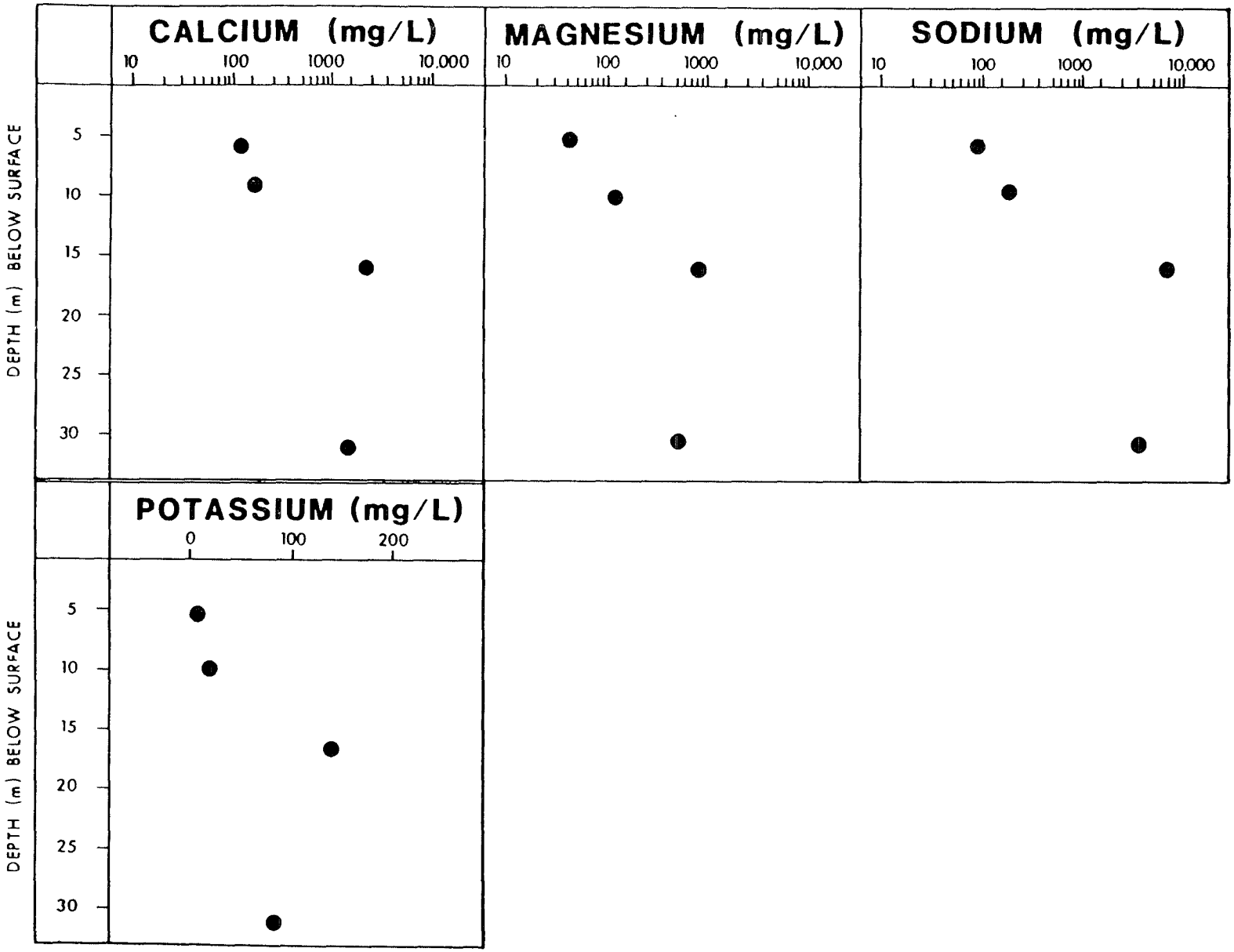


DRAWING No **19d**

HC9-85

WATER CHEMISTRY PROFILE B - CATIONS

SITE F

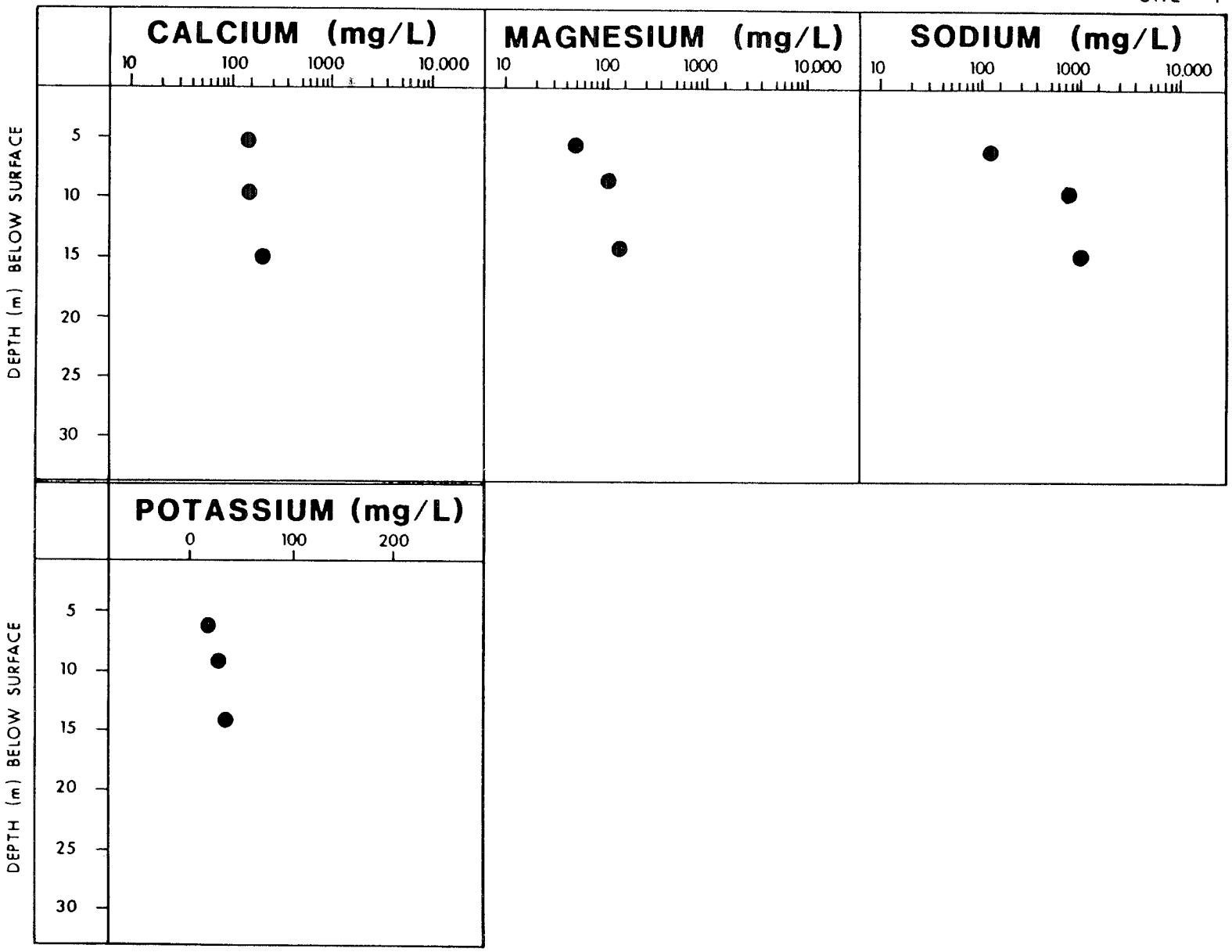


DRAWING No **19e**

HC10-85

WATER CHEMISTRY PROFILE B - CATIONS

SITE F

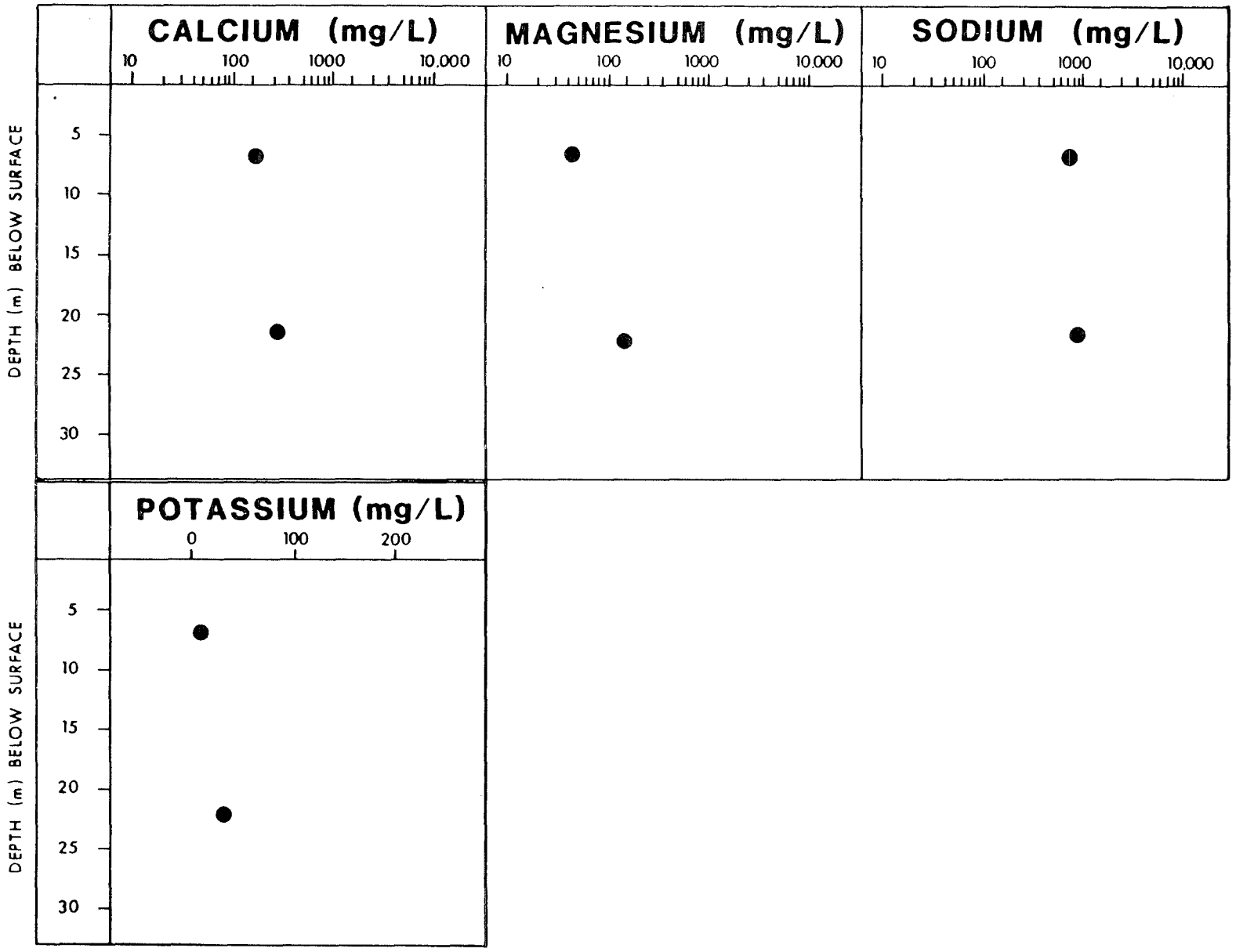


DRAWING No 19f

HC11-85

WATER CHEMISTRY PROFILE B - CATIONS

SITE F

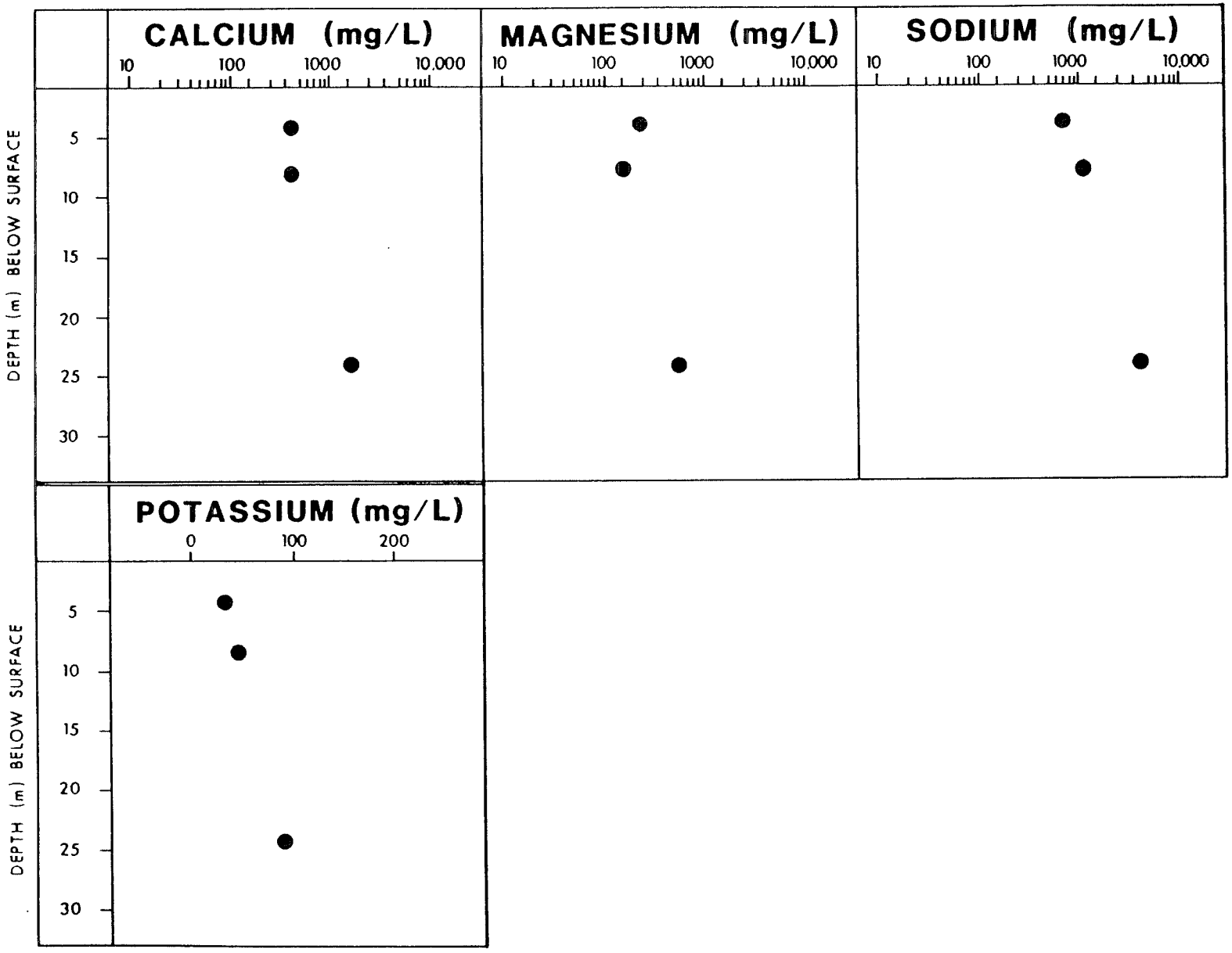


DRAWING No **19g**

HC12-85

WATER CHEMISTRY PROFILE B - CATIONS

SITE F

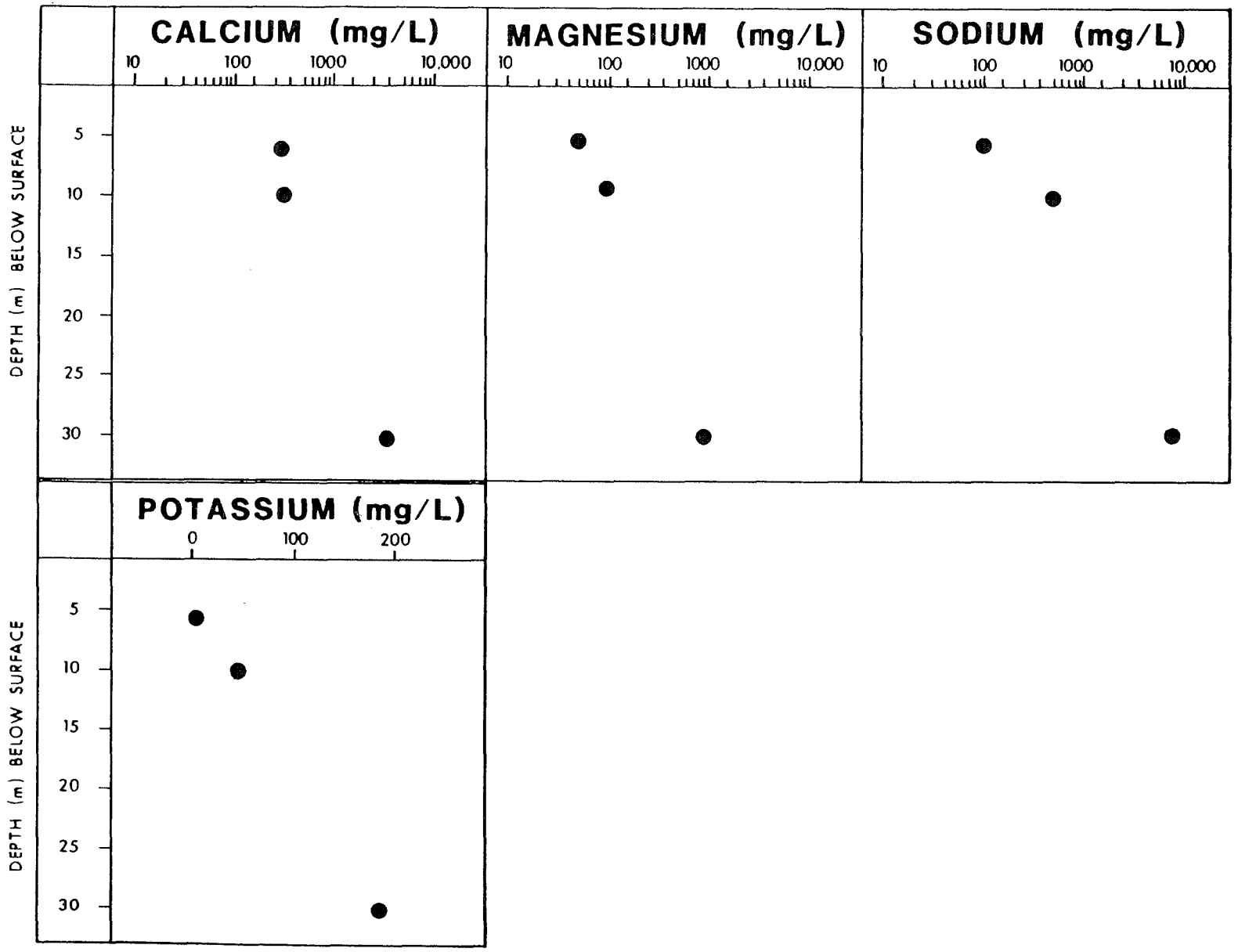


DRAWING No **19h**

HC13-85

WATER CHEMISTRY PROFILE B - CATIONS

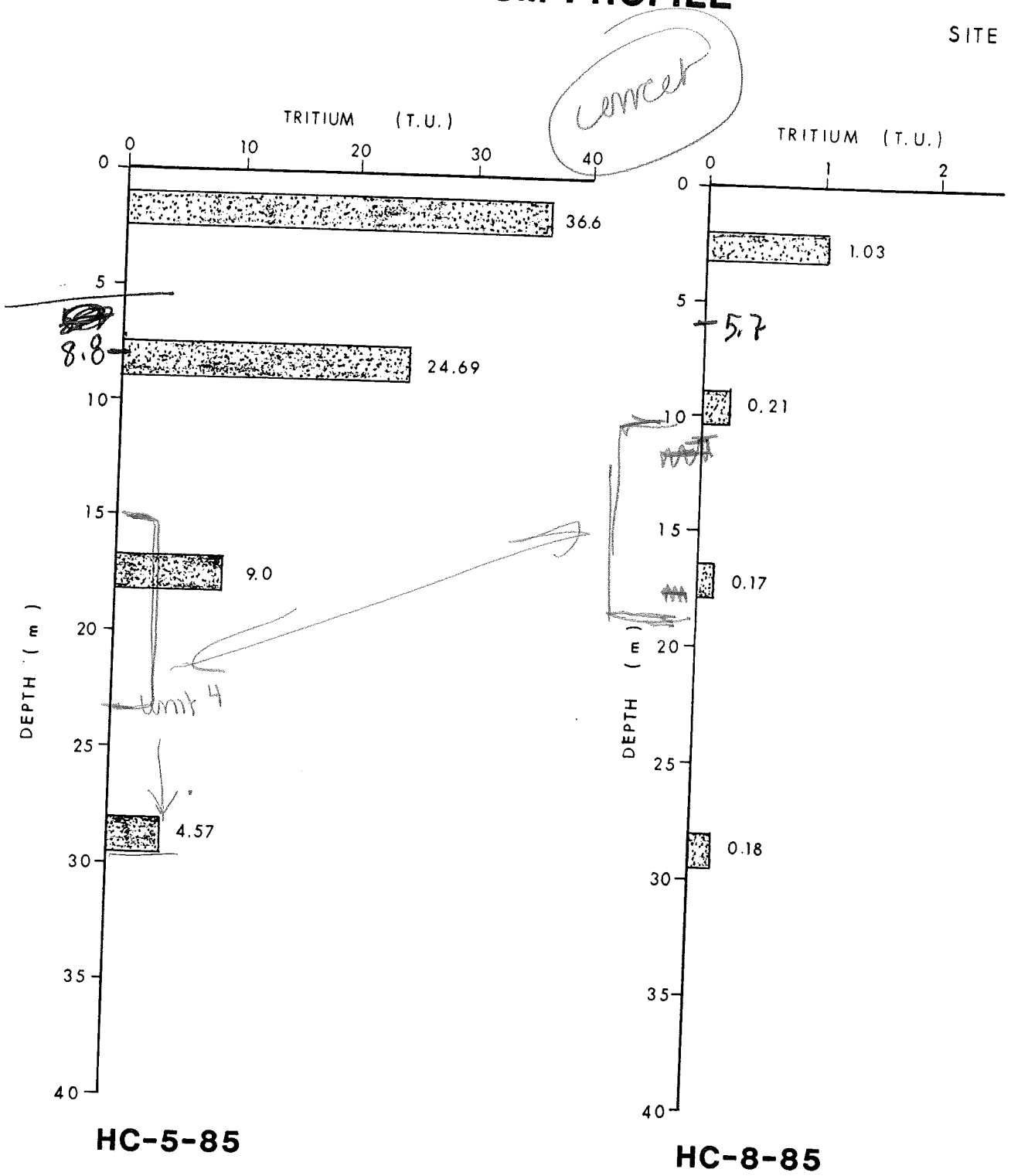
SITE F



DRAWING No 19i

TRITIUM PROFILE

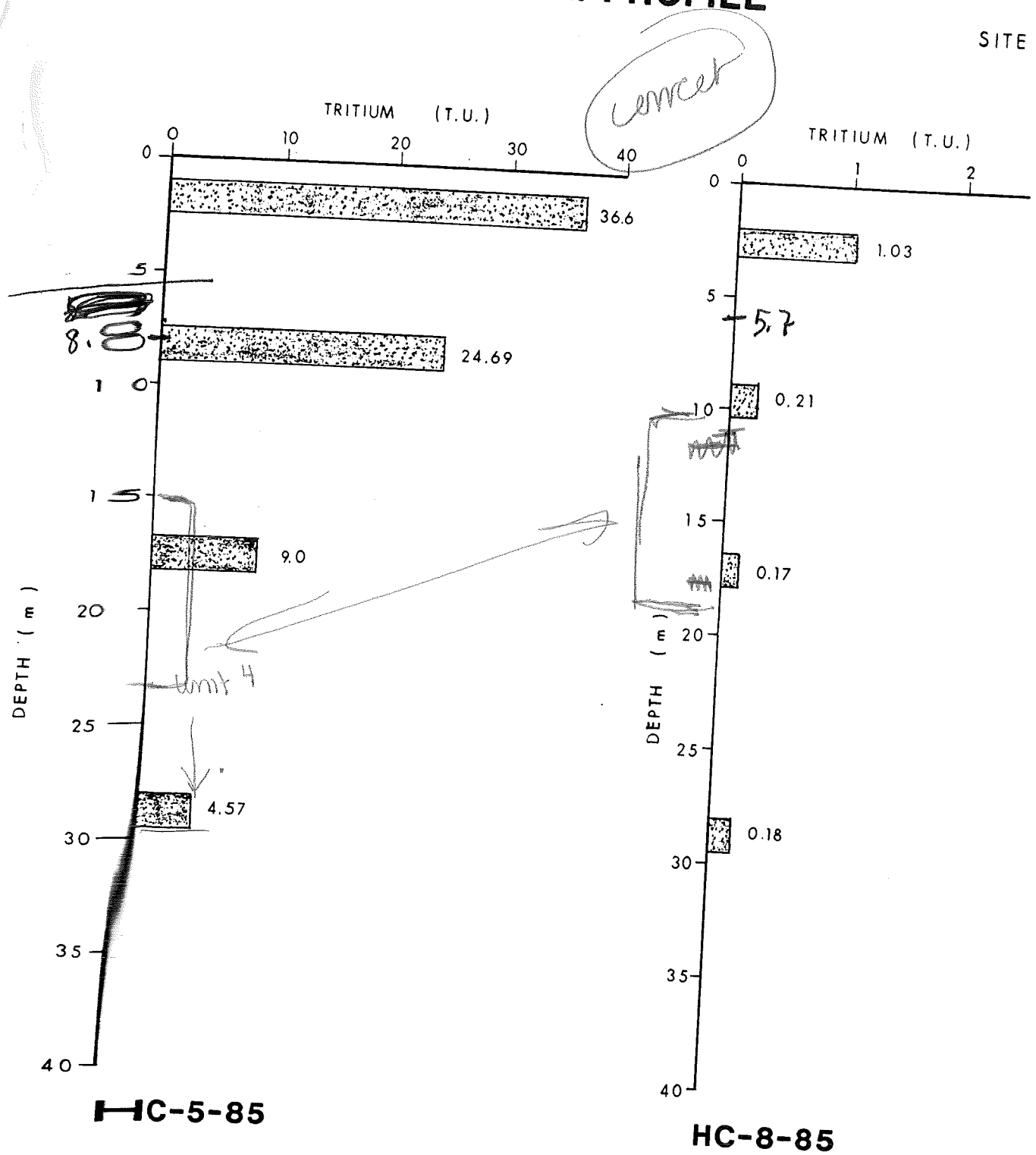
SITE F



Drawing No. 20

TRITIUM PROFILE

SITE F



Drawing No. 20