HYDROGEOLOGICAL INVESTIGATIONS AT THE BAYVIEW PARK LANDFILL BURLINGTON, ONTARIO

Progress Report

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INTRODUCTION

In the past several years, research by the University of Waterloo on the movement of landfill-derived contaminants has been primarily restricted to unconfined sand aquifers with relatively simple stratigraphy and groundwater flow patterns. The major processes working to produce contaminant plumes in this type of geologic setting are now beginning to be understood, and so research efforts are expanding to investigate contaminant movement in groundwater at landfill sites situated on fractured rock. As part of this effort, a study began at the Bayview Park Landfill in the spring of 1982.

The Bayview Park Landfill is located in Burlington, Ontario about 4 2 km north of Lake Ontario on the flank of the Niagara Escarpment (Figure 1). There are two main objectives of the hydrogeological investigations at the site. These are:

i) to delineate a zone of groundwater that has been impacted by the landfill. The shape and position of this zone will then be compared to the position predicted by conventional methods based on equivalent porous medium concepts in order to determine the applicability of these methods to fractured rock environments;

ii) to identify any major mechanisms of attenuation and the degree to which they may be effecting contaminant migration in the groundwater system in the fractured bedrock.

In pursuit of these objectives, detailed hydrogeologic data collected from multilevel devices and conventional piezometers will be used to examine the groundwater flow system and water quality trends in the vicinity of the site.

The purpose of this report is to describe the groundwater monitoring system that now exists at the site along with the results of groundwater sampling between May, 1982 and February, 1983. At this stage in the investigation, the distribution of diagnostic contaminants is not yet mappable so the information presented herein must be considered preliminary. The results of an additional sampling program, conducted during March, 1983 will be presented in a later report.

METHODS OF INVESTIGATION

The monitoring network at the site consists of 33 conventional piezometers that are operational. These piezometers are generally located in nests of three. The monitoring network also includes seven multilevel devices of the type described by Cherry and Johnson (1982). These devices contain several piezometers, each connected to a discrete vertical interval. The conventional piezometer nests and multilevel devices together provide a total of over 100 locations across the study site where hydraulic head measurements and groundwater samples can be taken.

The conventional piezometers were installed between 1979 and 1981 by private hydrogeological consultants (Morrison Beatty Limited). The majority were placed in 13 cm (5 inch) diameter augered boreholes. The screened intervals were surrounded by a sand/gravel filter and sealed from the rest of the borehole by a bentonite plug.

The seven multilevel devices, identified as the WAT series in Figure 2, were installed in the vicinity of the landfill during the summer of 1982 to compliment the existing monitoring network. These devices have recently been developed specifically for groundwater monitoring in fractured rock and have several advantages over conventional piezometer nests (Cherry et al., 1983). The latter requires that each piezometer be placed in a separate borehole. The multilevel device, however, allows several discrete fractures or fracture zones to be isolated within a single borehole. Single piezometer tubes extend to the ground surface from each of these zones for the purpose of monitoring (Figure 3). A multilevel device can therefore provide a much more refined vertical distribution of contaminants while minimizing drilling costs.

The first of these devices, which was an experimental prototype, was installed in a 7.5 cm diameter borehole to a depth of 60 m. The borehole was cored at WAT 1 (Figure 2) to determine the thickness of the refuse and to examine the stratigraphic sequence and fracture network in the bedrock beneath the study site. Five sampling intervals, each about 0.6 m in length, were selected for monitoring from an examination of the rock core.

The other multilevel devices were installed in 11 cm diameter boreholes created by an air-tricone method of rotary drilling without coring. This method allowed drilling to proceed much more quickly without the use of significant amounts of drill water. The use of drill water should be avoided because it must be removed before representative groundwater samples can be obtained.

From an examination of the core taken at WAT 1, it was assumed that fractures occur at all depths in the bedrock. As a result, certain modifications were made to the multilevel device. The larger borehole diameter created by the air-tricone method permitted the number of sampling intervals to be increased to between 10 and 12 and the actual interval length to be increased to 1.2 m. Because the total depths of these boreholes were to range from 35 to 45 m, these changes allowed most of the borehole to be monitored thereby maximizing the possibility of intersecting fractures through which contaminant migration may be occurring.

At WAT 2, the multilevel device was installed to a depth of 46 m and contains 10 sampling intervals. The remaining devices each contain 12 sampling intervals. WAT 3 was installed to a depth of 39 m and WAT 4, 5, 6 and 7 were all installed to a depth of 37 m. The cross-sectional distribution of these sampling intervals in the vicinity of the landfill is shown in Figure 4. A detailed discussion of the construction and installation of the multilevel devices is given by Cherry et al. (1983).

After installation, a fluoranine dye tracer was used to test all of the devices for leakage. It is important to identify any connection between the inside of the device and the actual sampling interval which may introduce uncertainty in hydraulic head measurements and the integrity of groundwater samples. Leakage effects were identified in some piezometers in most of the multilevel devices, however these effects will be eliminated when a bentonite slurry is injected to completely seal the devices in the spring of 1983.

The entire monitoring network was thoroughly developed prior to any groundwater sampling. All of the conventional piezometers were bailed dry and the water levels allowed to fully recover. This procedure was only done once because nearly all of these piezometers had been bailed out on several occasions during earlier investigations conducted by a private consultant. Several standing volumes of groundwater were pumped from each piezometer tube in the multilevel devices using a triple-tube gas drive system as described by Robin et al. (1982). The chloride and total organic carbon concentrations of this groundwater were determined on a regular basis. Pumping of the multilevel devices was done on several occasions until the values of these two parameters stabilized, indicating that the effects of the drilling activity on the groundwater in the sampling zones had been minimized.

After all the monitoring devices had been thoroughly pumped out, several water-level measurements were taken in order to monitor hydraulic recovery and to establish the equilibrium hydraulic head distribution across the site. The measurements were taken with a Roctest narrow diameter electric probe with accuracy to within $\pm$ 2 cm.

Rising-head tests were conducted on the piezometers in the multilevel devices to determine hydraulic conductivity. This required monitoring water level recovery in a piezometer tube following evacuation of the standing water volume. The recovery versus time data was then used in the conventional Hvorslev method to determine fracture permeability of the bedrock. The results of these tests will be discussed later in the report.

Groundwater from ten of the conventional piezometers was sampled for enriched tritium analysis by a liquid scintillation technique. The enriched method provides results with an accuracy of $\pm$ 1 TU. One litre samples were collected with a stainless steel bailer thoroughly rinsed in the sample water and shipped to the Lawrence Livermore Laboratory at the University of California for analysis.

PHYSICAL SETTING

The Bayview Park landfill is located in the City of Burlington, Ontario on the edge of the Niagara escarpment that slopes downward toward Lake Ontario (Figure 2). It occupies approximately 18 hectares and was in operation between the early 1960's and early 1970's. The refuse received at the site was largely municipal waste from the City of Burlington along with some industrial waste. It was deposited in a depression made by shale quarrying operations in the area.

Surface elevations across the study site reflect the slope of the escarpment toward the southeast. Elevations are approximately 160 m (525 ft asl) in the vicinity of the landfill but decrease to approximately 122 m (400 ft asl) near the North Service Road (Figure 2).

The site is dissected by several small ravines oriented from northwest to southeast. Intermittent creeks, generally dry during the summer months, run through these ravines toward Lake Ontario. The largest of these emanates from the toe of the refuse and flows through a narrow marsh that extends almost to the North Service Road and remains wet year round.

The overburden at the site is typical glacial drift containing a significant amount of silt and clay-sized material imported from the underlying shale bedrock (Chapman and Putnam, 1966). It is thickest northwest of the landfill in the vicinity of WAT 2. Elsewhere across the site the drift is generally only a few metres thick or absent altogether.

The bedrock beneath the glacial drift is a red shale of the Queenston Formation, Upper Ordovician in age and likely deposited in a 'deltaic' estuarine environment (Liberty and Bolton, 1971). It is essentially continuous to a depth of about 135 m at the site and is underlain by grey carbonates of the Georgian Bay Formation. Occasional grey green shaly limestone bands were observed in the rock core however these rarely exceeded 0.2 m in the thickness. The top few meters of the shale are extremely weathered. Below this, the shale has an extensive fracture network that appears to have developed mainly along horizontal bedding planes. Because the intact bedrock does not have any significant permeability it is reasonable to assume that groundwater moves primarily through the existing network of horizontal fractures.

RESULTS AND DISCUSSION

Physical Hydrogeology

Water level measurements were collected on several occasions for the entire monitoring network and used to characterize the groundwater flow system at the site. The horizontal component of the hydraulic gradient measured between the shallowest piezometers at locations OW 1 and OW 13, is toward the southeast at a gradient of 0.044. The vertical component of the hydraulic gradient was determined from differences in hydraulic head between pairs of piezometers in the conventional well nests and the multilevel devices. Vertical profiles of hydraulic head for WAT 1 - WAT 7 are shown in Figures 6 to 11.

The hydraulic head profiles for most of the multilevel devices are quite irregular and contain both upward and downward differentials which in some cases are quite large. These trends are probably related to the bedding features and associated fracture permeability of the shale. The distinct differences in head for each piezometer also indicate that the individual sampling intervals are effectively isolated by the seals or packers and the head measurements in the piezometer tubes are not being effected by hydraulic connection with other sampling intervals.

WAT 1 is situated near the downgradient edge of the landfill (Figures 2 and 4) and contains the deepest sampling intervals across the study site. All of the piezometer points are at least 30 m beneath the landfill surface. At these depths the hydraulic head differentials are all upward ranging from 0.7 to 7.7 m. These points may intersect the regional flow paths discharging to Lake Ontario.

WAT 2 was installed upgradient of the landfill to provide background water quality. Vertical differentials between pairs of piezometers in this multilevel device range from a minimum of 0.2 m to a maximum of 5 m. At depths shallower than about 30 m below ground surface these head differentials are downward; at greater depth they are upward.

The remaining multilevel devices are all located downgradient of the landfill. WAT 3 is about 30 m from the refuse limit. Hydraulic head differentials range from 0.15 to 1.2 m but the profile is very irregular and does not exhibit any general upward or downward trend. The profile for WAT 4 however, does indicate a gradual decline in hydraulic head with depth. These downward differentials range from 0.3 m to 2.3 m.

At WAT 5, about 140 m from the landfill, the hydraulic head profile is much less irregular and differentials do not occur over as wide a range as in the other multilevel devices. At shallow depth, flow is generally downward. Between 12 and 18 m below ground surface the hydraulic head elevation drops 4 m and remains low in all of the piezometers below this depth. The differentials for this device range from a minimum of only 0.1 m to a maximum of 1.4 m.

At WAT 7, the hydraulic head profile indicates mostly downward differentials between pairs of piezometers, generally ranging from 0.12 m to 2.2 m. At a depth of about 23 m there is a significant change in the profile similar to that shown for WAT 5. The hydraulic head elevation decreases by 6 m, below which the heads show minor fluctuation. This significant drop may indicate a more permeable stratum that is acting as a "deep hydraulic drain" in the system.

The generally downward differentials between pairs of piezometers described for the individual multilevel devices appear to reflect the regional downward flow of groundwater south of the landfill. The irregular upward and downward trends in the vicinity of WAT 4 and OW 11 may be related to the position of these wells in a topographic low and very wet area. The hydraulic head measurements at shallow depths may be influenced by a very local flow system that discharges to a marsh in this vicinity.

Hydraulic conductivity values for the shale were determined by rising-head tests conducted on the piezometer tubes in each multilevel device. The time required for complete recovery of the water level after evacuation of the standing volume ranged from several minutes to several hours. This range is to be expected considering the arbitrary isolation of fracture intervals in the borehole during installation of each multilevel device. The relatively fast response times indicate a sampling interval which intersects several fractures or a few large fractures while the slower response times indicate very few or very small fractures.

The standard Hvorslev method (1951) was used to analyse the data. With this method a basic time lag is determined from a plot of relative water level recovery versus time. In theory this plot should be linear. For the majority of tests conducted in the shale however, this plot was non-linear with a typically steep slope during early times and a much more gradual slope at later times (Figure 5). This two-stage response may be the result of an outward decrease in hydraulic conductivity of fractures that connect to the sampling interval or it may be due to the termination of fractures in the shale near the sampling interval.

Hydraulic conductivities representative of those found at the site are given in Table 1. They were determined using the formula

$$K = r^2 \ln (L/R)/(2\ LT_0)$$

where r is the radius of the piezometer, L is the length of the intake zone, R is the radius of the intake zone and T_0 is the basic time lag. For the piezometers that responded with a distinctly non-linear plot of relative water level recovery versus time, two values of T_0 were used in the calculations. The first value was taken from the steep portion of the curve during early time and the second value was taken from the more gradually sloping portion of the curve at later times after it was shifted to pass through the origin.

The two time lag values therefore generated two values of hydraulic conductivity for most piezometers. These ranged over two orders of magnitude from 10^{-5} cm/s using the early time lag to 10^{-7} cm/s using the later time lag. The latter is probably a more representative bulk hydraulic conductivity for the shale because it represents the characteristics of recovery over a larger rock mass. The geometric mean of the hydraulic conductivities calculated using the later time lag values is 4.57×10^{-7} cm/s, which indicates that the shale has a slight to moderate fracture permeability.

This hydraulic conductivity must be considered to apply only in the horizontal plane because of the primarily horizontal orientation of the

TABLE 1

Representative hydraulic conductivities for the Queenston shale in the vicinity of the Bayview Park Landfill

Location	Sample Point	Depth (m)	K (cm/sec)
WAT 3	4	17.5	*(i) 2.9×10^{-5}
			(ii) 4.4×10^{-7}
	5	20.1	(i) 3.0×10^{-5}
			(ii) 5.1×10^{-7}
	6	22.7	(i) 6.6×10^{-6}
			(ii) 2.4×10^{-7}
	10	33.2	(i) 2.3×10^{-7}
WAT 5	1	6.4	(i) 2.1×10^{-5}
		22.2	(i) 2.3×10^{-5}
			(ii) 8.0×10^{-7}

* (i) T_o taken from early time

(ii) T_o taken from late time

bedding planes and the isolation of strictly vertical segments of the borehole by the multilevel devices. Multilevel devices installed in angle boreholes would be required to evaluate vertical hydraulic conductivity at the site.

An estimate of the flux of leachate entering the groundwater system is desirable in this type of hydrogeological investigation, however at this stage estimates of this kind are not reliable. This is because the vertical hydraulic conductivity of the shale beneath the landfill and the infiltration characteristics of the landfill cover are both unknown. As indicated above, the rising-head tests only provide an estimate of horizontal hydraulic conductivity in the area downgradient of the landfill.

Average linear groundwater velocity is an important parameter in tracking the movement of landfill-derived contaminants through the system. It is calculated by the expression:

$$v = ki/n$$

where k is the hydraulic conductivity, i is the hydraulic gradient and n is porosity. In unconsolidated porous media, porosity represents the volume occupied by the intergranular void spaces relative to the total volume of the porous media. In fractured rock however, porosity represents the volume occupied by the fractures through which groundwater flows relative to the total volume of the rock mass. Porosity of fractured rock is generally 2 or 3 orders of magnitude smaller than in granular deposits and it is considerably more difficult to accurately measure. There are no fracture porosity values for shale cited in the literature, however for the purpose of illustrative calculations in this report, a porosity of 1 x 10^{-3} will be used along with the geometric mean of late time hydraulic conductivities and the lateral hydraulic gradient across the site measured between OW 1 and OW 13. This produces an average linear groundwater velocity through the fractured shale of approximately 6 m/yr. Moving at this rate, groundwater in the vicinity of the landfill when it began operation about 20 years ago would currently be situated about 150 m downgradient. Contaminants moving into the groundwater at this time may also have travelled this distance if they have not been retarded in some way in the shale environment.

ie 600years.

At the Bayview site, this average linear groundwater velocity should only be used in order to get a rough indication of where contaminants might be in the system. There are large uncertainties involved in this calculation which prohibit its use for obtaining specific estimates of the position of the front of a zone of contaminated groundwater. The main uncertainties pertain to the unknown magnitude of the fracture porosity for the shale. Additional uncertainty relates to the wide range of values of hydraulic conductivity that probably exist for various zones in the system.

uncertainties

In order to illustrate the magnitude of uncertainty associated with these numbers, consider the effect of changing the hydraulic conductivity. Values of hydraulic conductivity on the order of 10^{-5} cm/s were generated from early time water-level-recovery data. While this value may not be truly representative of the bulk capability of the shale to transmit groundwater, it does indicate that there are individual fractures in the rock that are able to yield water at this must faster rate. The potential therefore exists for landfill-derived contaminants to move into one of these fractures and be transported downgradient at a much faster rate than the average hydraulic conductivity used in previous calculations. The geometric mean of hydraulic conductivities determined from early time water level data is 9.29×10^{-6} cm/s. This value yields an average groundwater velocity of 130 m/yr. Using the rate of transport, contaminants may have reached Lake Ontario while the landfill was still in operation.

The problems involved with quantifying groundwater flow at the Bayview site outlined in this section obviously make it essentially impossible to predict the extent of contaminant migration from the landfill. Rather, the indications provided by the basic equations will have to be substantiated by mapping of diagnostic landfill contaminants that have travelled through the system to their present location. This representation of contaminant distribution will provide an appreciation of the uncertainties and limitations of the basic quantitative approach in a fractured shale environment. The identification of diagnostic contaminants and use of these contaminants to determine the size of the zone of contaminated groundwater is currently underway.

Water Chemistry

Between May and August, 1982, groundwater pumped from each piezometer tube to minimize the effects of drilling was regularly analyzed for chloride and total organic carbon. Following this, all the piezometers were sampled at least twice to establish vertical concentration profiles for both parameters. These profiles are shown in Figures 6 to 11.

Chloride and total organic carbon were the two parameters selected for analysis because they have been used successfully as conservative tracers for landfill contamination at several other sites in Ontario. At these sites background concentrations of both parameters were relatively low so that elevated concentrations contributed by the landfill provided quite distinct plume definition.

Sampling results for the Bayview site indicate that these parameters are not well suited for use as natural tracers of landfill contamination in groundwater moving through the shale. This is because the natural chloride and total organic carbon concentrations in the groundwater are extremely high and exhibit order of magnitude changes between sampling intervals within a single multilevel device. These trends can be seen in the vertical concentration profiles for WAT 2, the device installed to provide background water quality for the site (Figure 7). Chloride concentrations range from under 100 mg/L to over 9,000 mg/L and total organic carbon concentrations range from approximately 300 mg/L to 8,000 mg/L. These large fluctuations make it impossible to establish a range of typically background concentrations relative to which the degree of contamination downgradient of the landfill can be assessed.

Concentrations in the other multilevel devices at the site are also very high and exhibit similar, and in some cases more extreme, fluctuations with depth. At WAT 5 chloride concentrations are generally between 1,000 and 2,000 mg/L but do reach as high as 8,000 mg/L. The TOC profile is very irregular with concentrations ranging from about 30 mg/L to almost 3,000 mg/L. Both profiles at WAT 7 show order of magnitude changes in concentration above and below which, values are quite consistent. Above about 45 m, chloride concentrations are typically 100's of mg/L; below this depth they jump to 1,000's mg/L. TOC concentrations are 10's of mg/L above a depth of about 90 m, below which they are 1,000's of mg/L.

The concentration profiles for WAT 3 and WAT 4 show the least fluctuation. At WAT 3, the Cl^- concentrations are generally above 20,000 mg/L. These values probably reflect the proximity of this device to the landfill. At WAT 4, 30 m downgradient, concentrations are an order of magnitude lower. There is no corresponding decrease in TOC concentrations between these two sites however. TOC values are generally about 1,000 - 2,000 mg/L throughout both profiles.

A comparison of Cl^- and TOC profiles across the site can only serve to indicate gross changes in water chemistry which may be related to the movement of contaminants from the landfill. To avoid the problems of interpretation caused by the natural variability in the background water, tritium, a radioactive isotope that is not effected by a changing geochemical environment was used as a natural tracer at the site. Tritium was introduced to the atmosphere during the period of intense bomb testing between 1953 and 1969 and entered the groundwater system via infiltrating precipitation. Since 1969, the tritium levels in rain and snow have declined but they remain much above the values that occurred prior to 1953. Because tritium moved downward through the soil virtually everywhere, it can be used to determine estimates of vertical movement rather than longitudinal movement of groundwater.

The results of the enriched tritium analyses are shown in Figure 12. The distribution of values follows the expected trend and is consistent with the distribution described previously by private consultants (Morrison Beatty Ltd., 1980 and 1981). The higher values at shallow depths are more characteristic of tritium levels in current day precipitation. The decreasing values with depth generally reflect radioactive decay of the tritium during the time it has been in the flow system and may also be due to a certain degree of tritium diffusion from the groundwater in the fractures into the intact shale blocks. At OW 12, this trend is reversed and a higher tritium value is found at depth. This is probably the result of some vertical fracture connections which have allowed more highly tritiated water in the shallow zone to move directly to much deeper zones.

The limit of the bomb tritium zone, as shown in Figure 12, was based on the enriched tritium data. At depths of about 23 m, values were approaching the detection limit so it therefore reasonable to assume that the maximum vertical extent of tritium in the groundwater probably exists at depths slightly greater than this.

The distribution of tritium in the system at the Bayview Park landfill does have significance in terms of identifying possible zones where landfill-derived contaminants may occur in the groundwater system. If tritium input to the groundwater zone since 1953 has reached depths of 23 m, it is therefore likely that contaminants introduced by the landfill early in its operation in the 1960's may also have moved downward to nearly this depth. This would only be the case for conservative, non-retarded contaminant species that exhibit diffusion characteristics similar to tritium, however.

On-Going Investigations

While tritium has provided an indication of the depth of active groundwater flow at the site, to date no diagnostic contaminant has been identified that can indicate the depth and extent of movement of contaminants in the shale. Because of the extremely poor quality of the background water at the site, it appears that the most suitable tracer would be one that does not occur naturally in the groundwater. We are currently investigating the use of synthetic organic compounds in this capacity. These compounds, particularly chlorinated hydrocarbons, do not occur naturally in the shale. They likely occur in the landfill, however. Some of these compounds may be mobile in the groundwater flow system in the shale and therefore they may serve as diagnostic parameters for plume mapping.

The use of synthetic organic compounds as leachate tracers is a relatively new concept in contaminant hydrogeology. While the analytical procedure to determine the presence of these compounds is now well developed, no efficient methods for routine sample collection in the field exist. Before the groundwater at the Bayview site can be tested to determine if a suitable organic tracer does exist, we are working to develop a reliable sampling device.

SUMMARY OF CONCLUSIONS

Hydrogeological investigations by the University of Waterloo have been on-going at the Bayview Park Landfill since the spring of 1982. Seven multilevel devices have been installed at the site, one to provide background water quality, the remaining six to monitor water quality with increasing distance downgradient of the landfill. Development and testing of these devices indicates that they are effective in isolating at least 10 vertical zones within a single borehole. Regular hydraulic head measurements and analysis of groundwater samples for chloride and total organic carbon for each of these discrete sampling intervals has established equilibrium profiles with depth.

The conventional approaches to describing groundwater flow have several associated uncertainties in the fractured shale environment at this site. In most cases, water level recoveries during rising-head tests produced two time lag values and therefore two hydraulic conductivities for each sample interval. These values were generally on the order of 10^{-5} cm/s for the early water level response and 10^{-7} cm/s for the late water level response. The geometric mean of all the hydraulic conductivities based on the later time water levels, 4.57×10^{-7} cm/s, was used in subsequent calculations because it was believed this value was more representative of the hydraulic properties of the shale on a larger scale.

The use of the average linear groundwater velocity to track the movement of contaminants is justified in a porous medium where hydraulic conductivity remains fairly constant and changes can easily be measured. The use of this parameter in a fractured rock environment however, is fraught with difficulty because the multilevel devices isolate discrete vertical intervals in the shale which generally contain a series of fractures rather than a single fracture. In this case then, field tests provide values for bulk hydraulic conductivity. Using the geometric mean of bulk hydraulic conductivities cited above yields an average linear groundwater velocity of about 6 m/yr.

While most of the bulk values were found to be on the order of 10^{-7} cm/s, the hydraulic conductivity of individual fractures probably ranges many orders of magnitude on either side of this value. In order to obtain an upper limit estimate of the rate at where contaminants may have moved, the average linear groundwater velocity was calculated using the geometric mean of hydraulic conductivities generated using early time water level response. This produced an average linear groundwater velocity of about 130 m/yr. Whether or not contaminants actually move at this rate cannot be determined with the existing data.

Although chloride and total organic carbon have been shown to be well suited for mapping contaminant plumes in unconsolidated sand aquifers beneath landfills in Ontario, they are not suitable to define the distribution of contaminants at the Bayview Park site. This is because the background quality water has very high concentrations of both these parameters and these concentrations show extreme fluctuations with depth. Without a relatively constant water quality against which other samples can be compared, it is essentially impossible to assess the degree of groundwater contamination related to the landfill. As a result, the Cl^- and TOC profiles are useful only to determine order of magnitude changes in water chemistry between the multilevel device sites and to monitor the stability of water chemistry with time at the individual sites.

Tritium was used as a natural tracer at the site because the time of input to the system is known and its concentrations are generally only effected by radioactive decay. Results indicate that tritium has penetrated to depths of at least 23 m. Considering that the beginning of tritium input to the groundwater system occurred not much earlier than the landfill operation, the tritium zone is also a zone where landfill contamination may occur.

In summary, the distinct differences in hydraulic head seen in the multilevel devices, the moderate permeability of the bedrock and the distribution of tritium all seem to indicate that the shale in the vicinity of the Bayview Park Landfill is a moderately active environment for contaminant transport. Investigations at the site are now focusing on the use of synthetic organic compounds as diagnostic tracers of landfill contamination. The distribution of these compounds will be used to confirm and refine the trends presented in this report and will aid in eventually developing a representation of the extent of landfill constituents.

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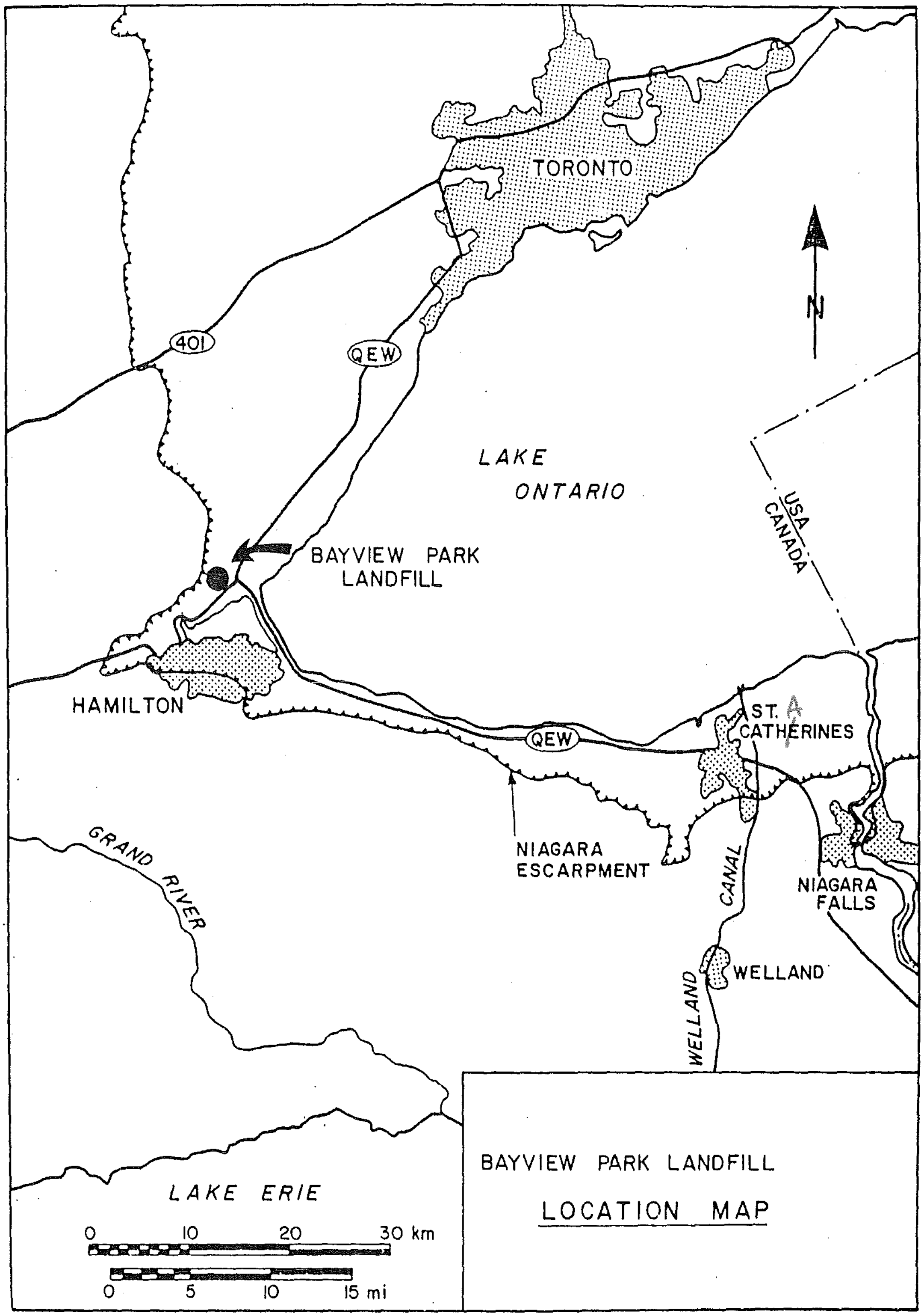

Figure 1. Location of the Bayview Park Landfill study site.

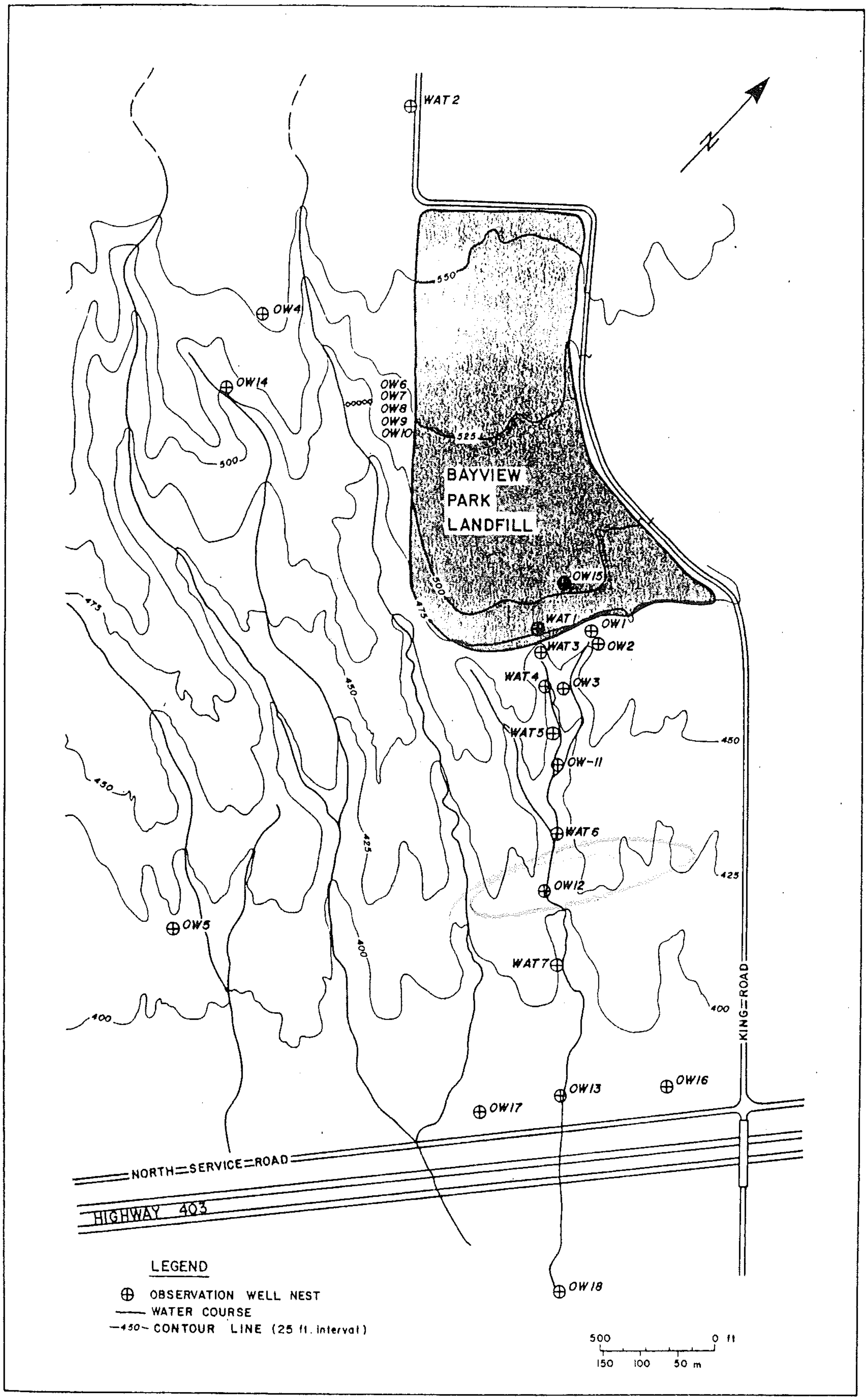

Figure 2. Plan map of the Bayview Park site showing surface drainage features and location of the groundwater monitoring network.

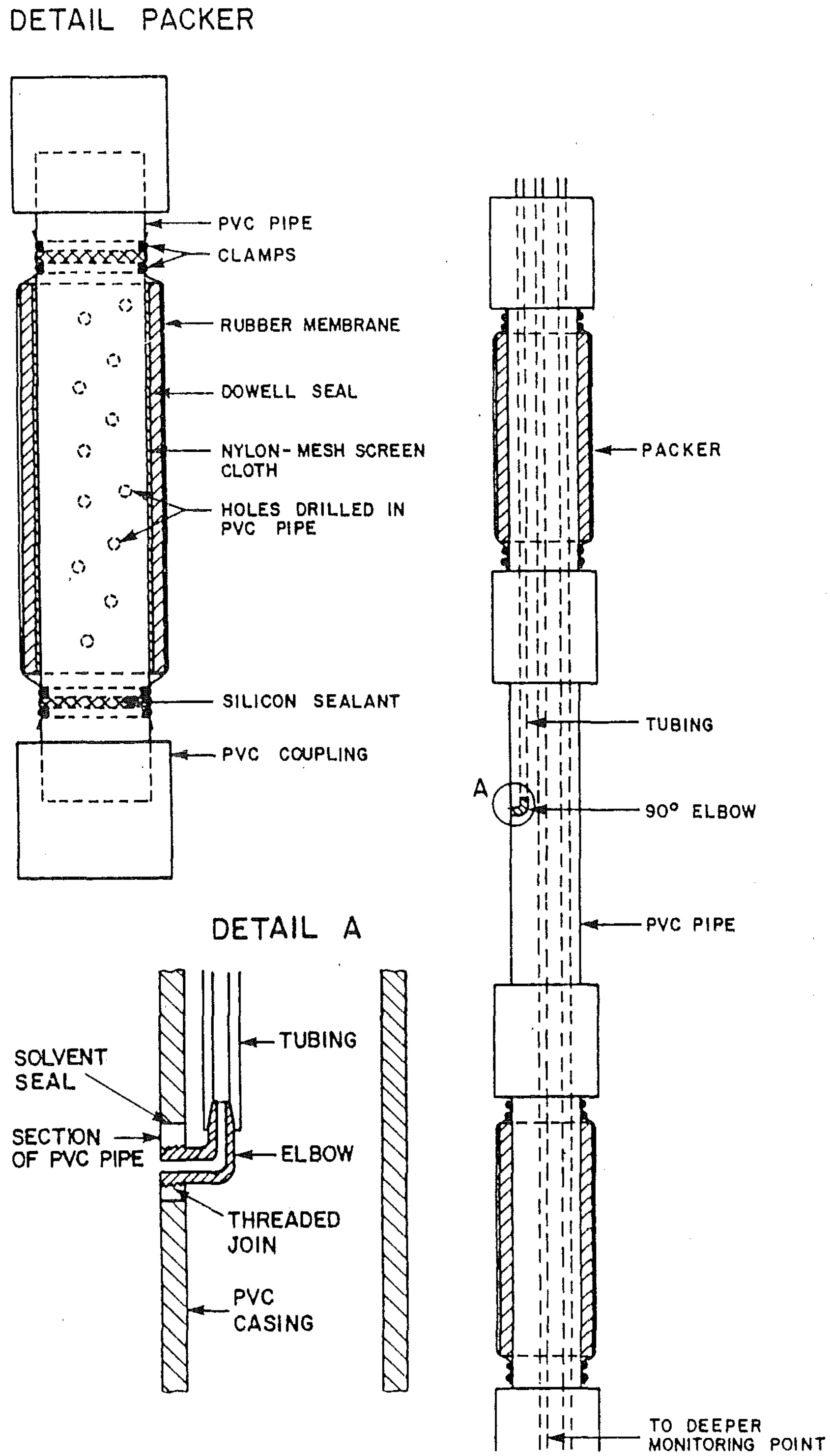

Figure 3. Schematic illustration of the components of the multilevel device (a) the packer system and (b) the monitoring port.

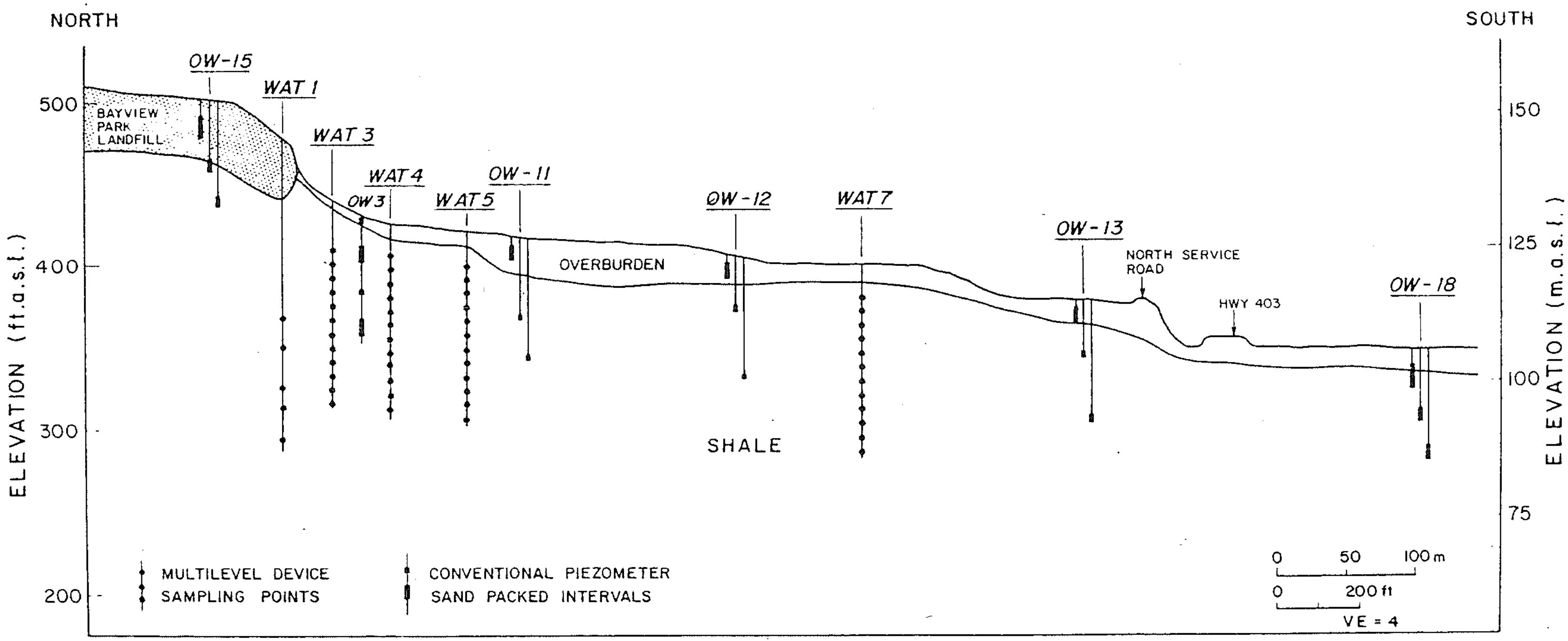

Figure 4. Hydrogeologic cross section along the main direction of groundwater flow at the Bayview Park site.

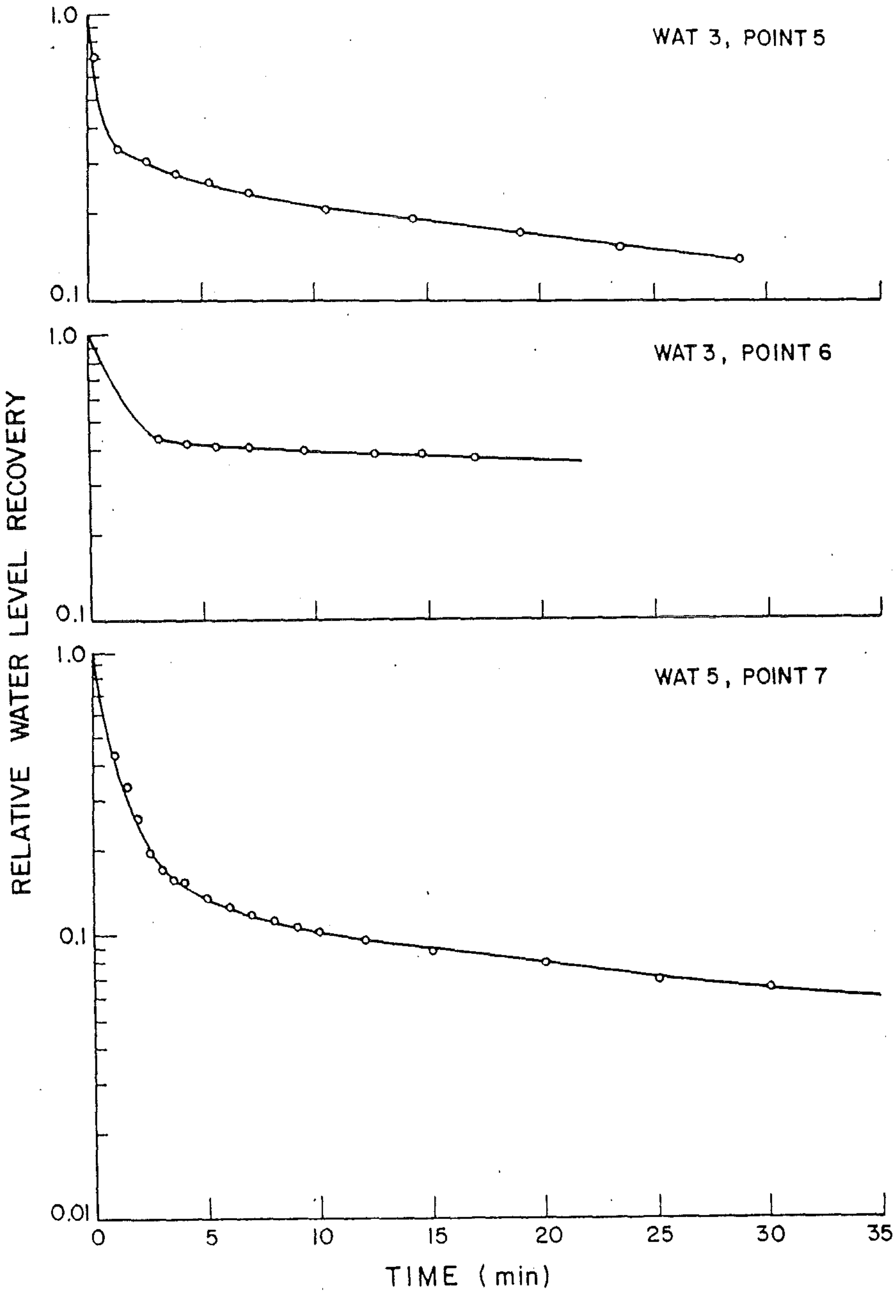

Figure 5. Typical Groundwater Response Curves.

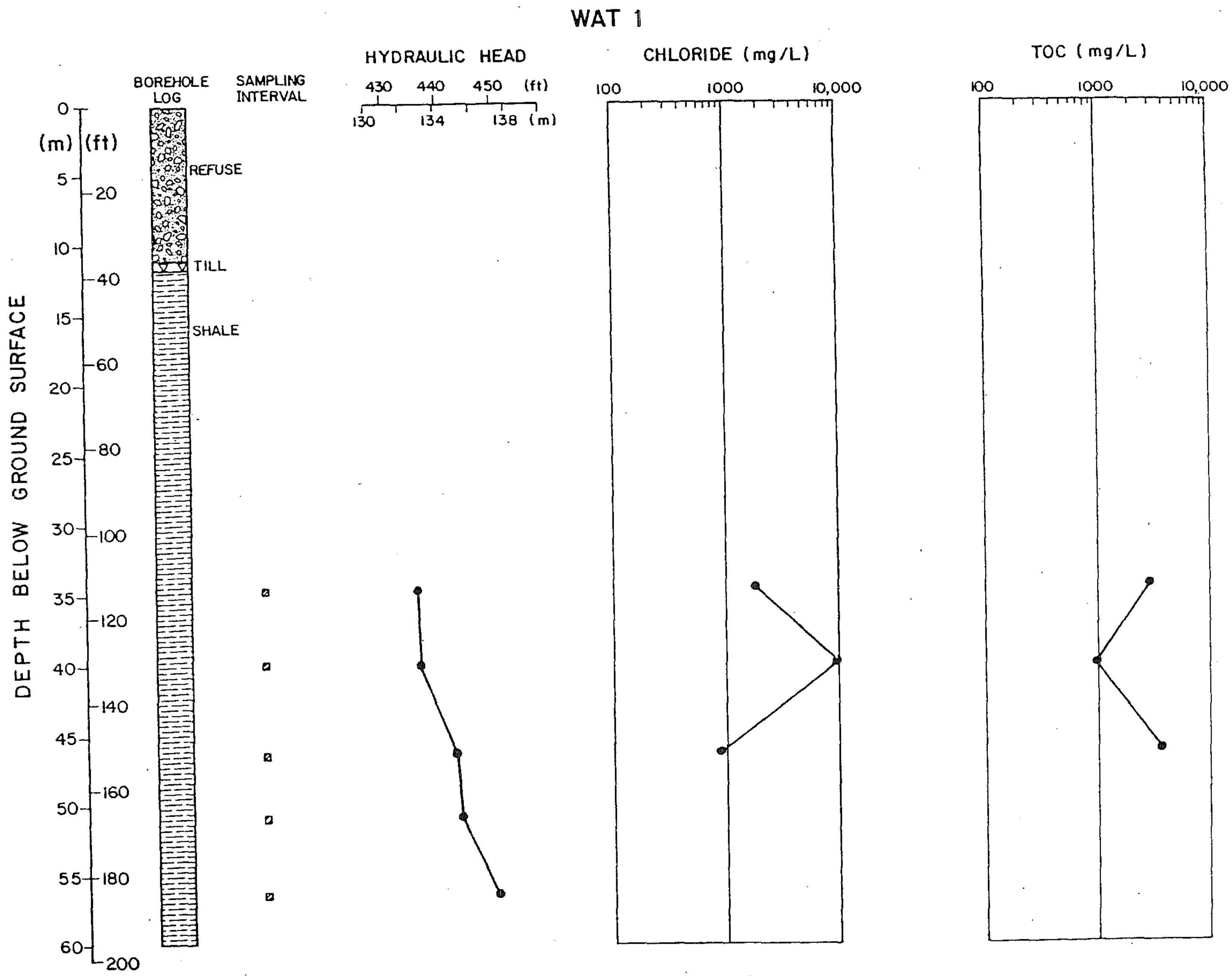

Figure 6. Depth profile of hydraulic head, chloride and total dissolved organic carbon for WAT 1.

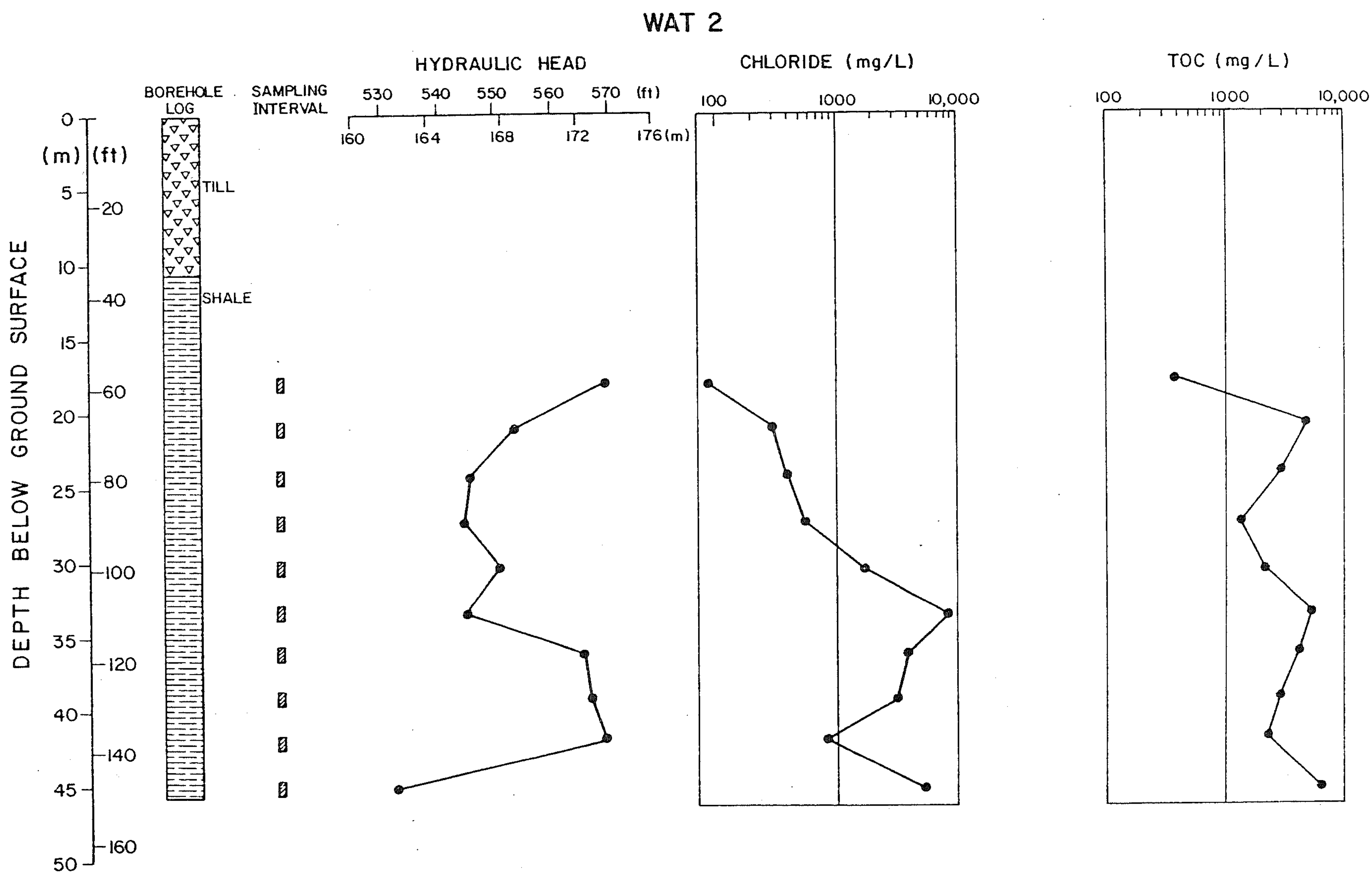

Figure 7. Depth profile of hydraulic head, chloride and total dissolved organic carbon for WAT 2.

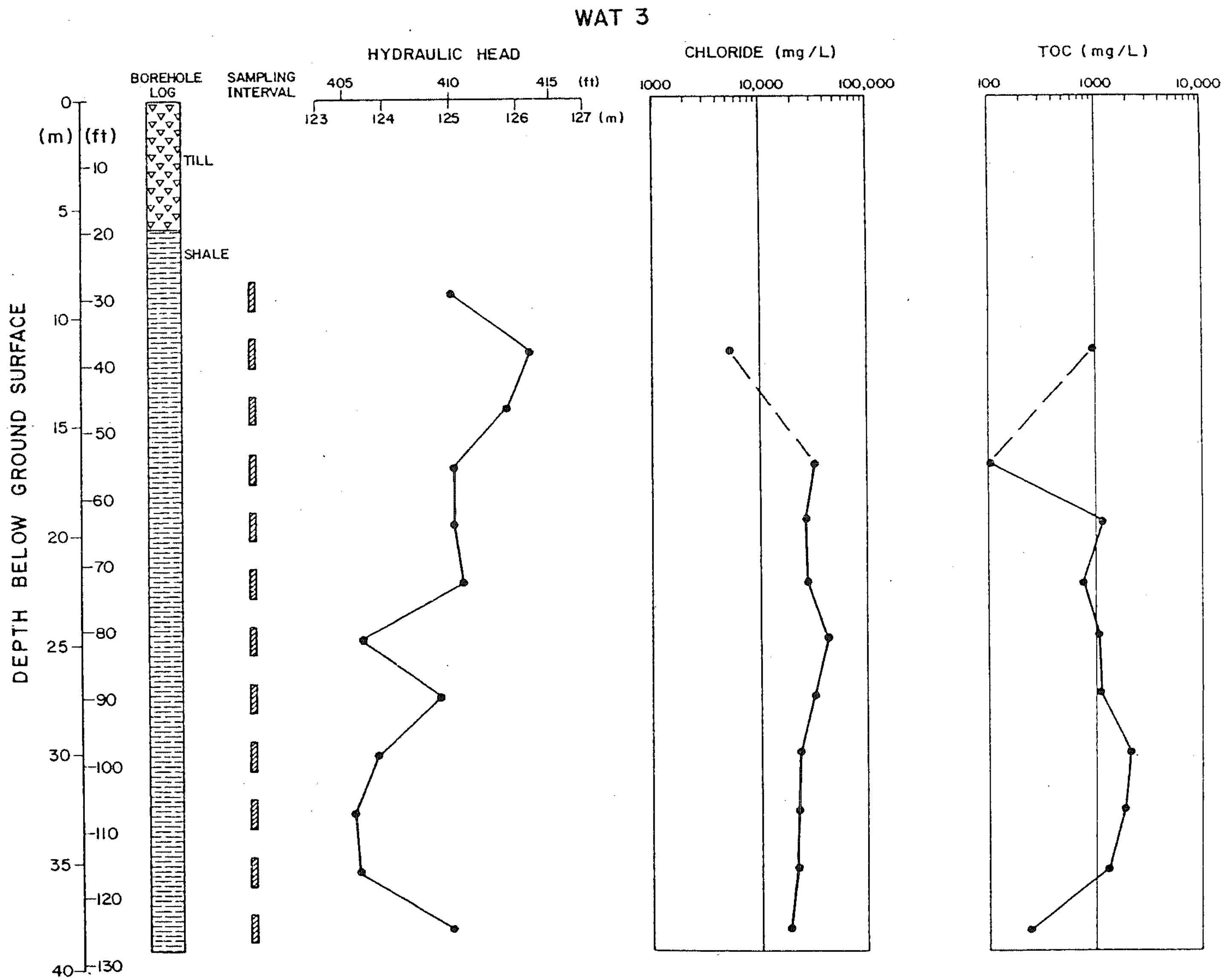

Figure 8. Depth profile of hydraulic head, chloride and total dissolved organic carbon for WAT 3.

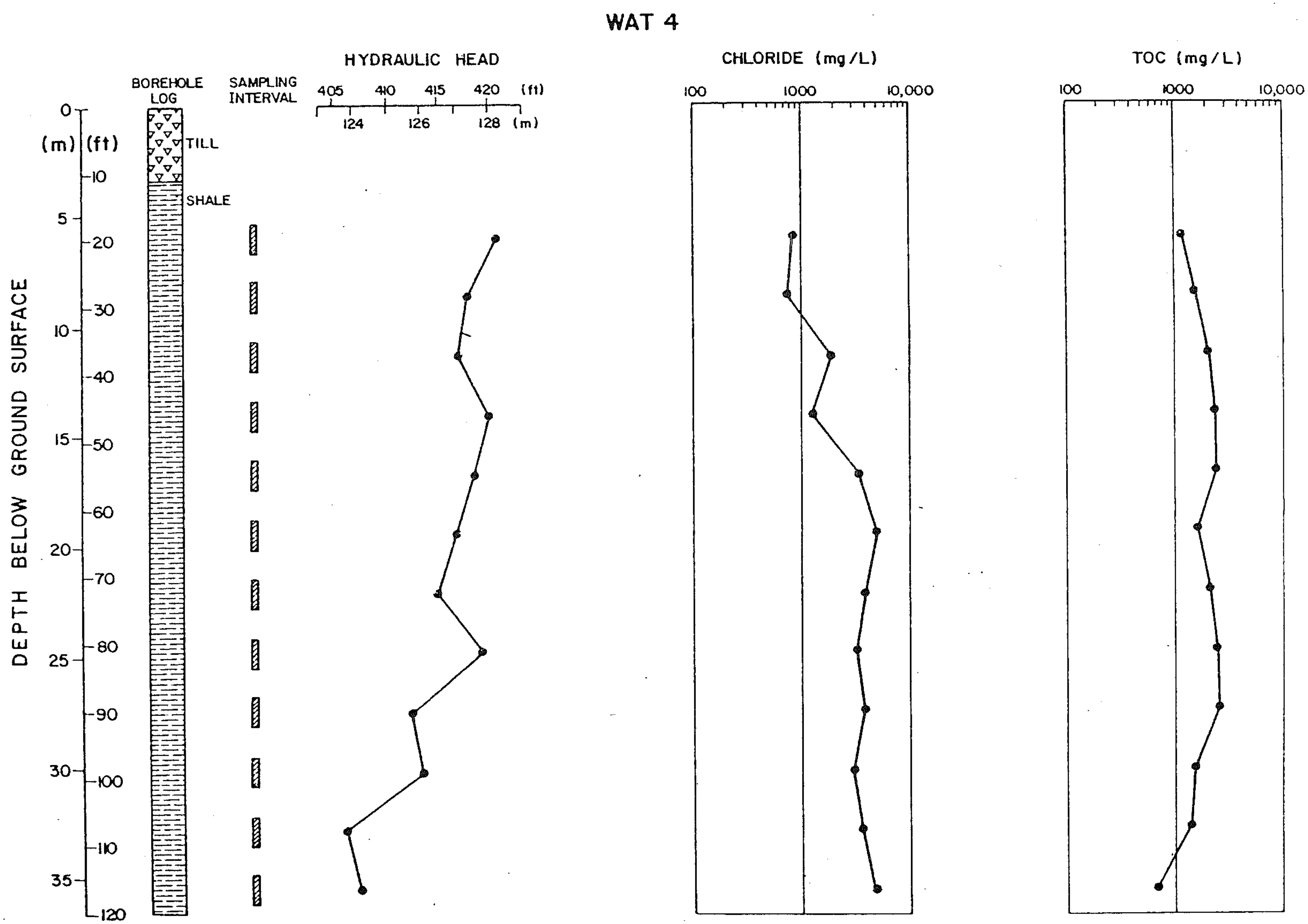

Figure 9. Depth profile of hydraulic head, chloride and total dissolved organic carbon for WAT 4.

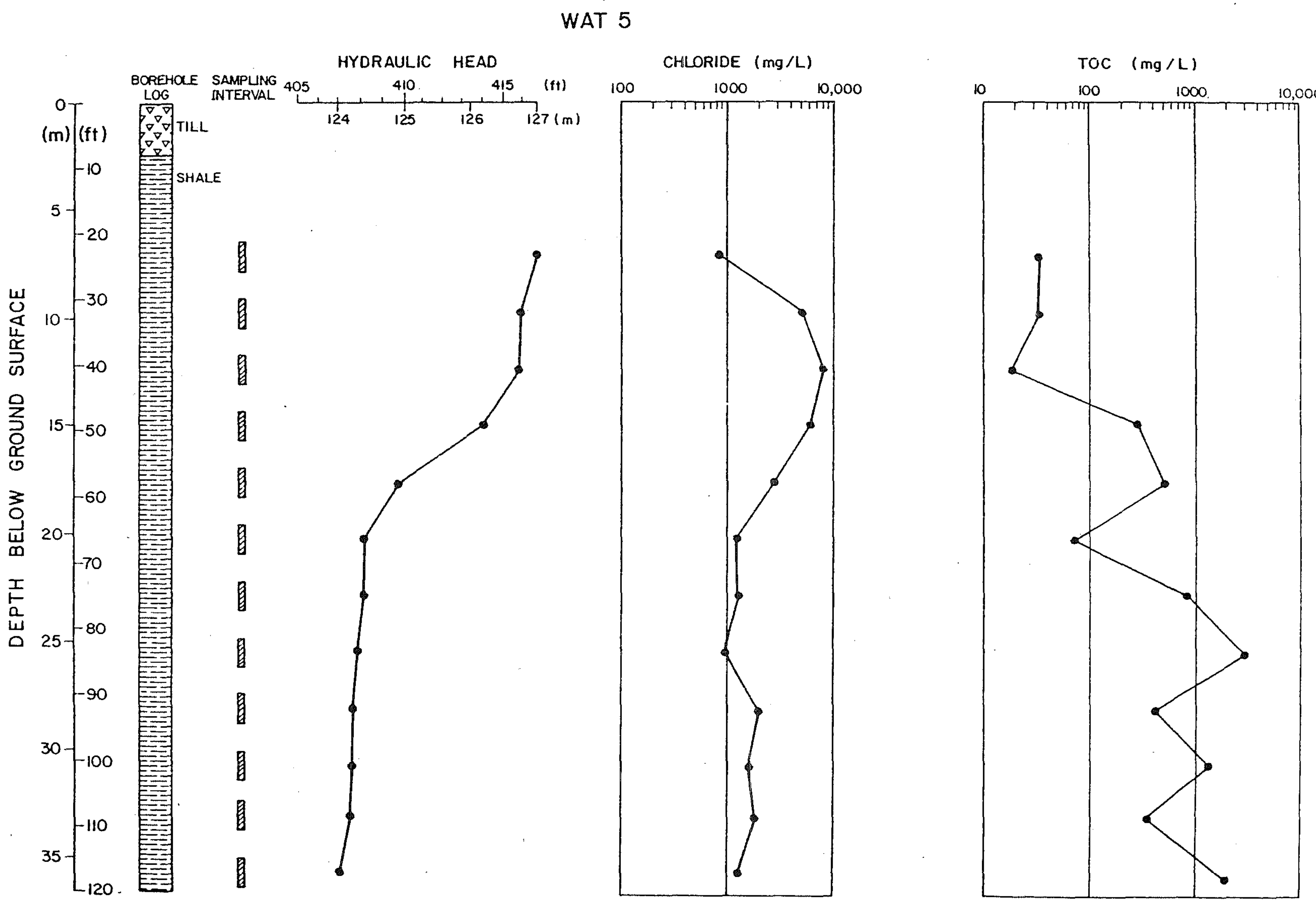

Figure 10. Depth profile of hydraulic head, chloride and total dissolved organic carbon for WAT 5.

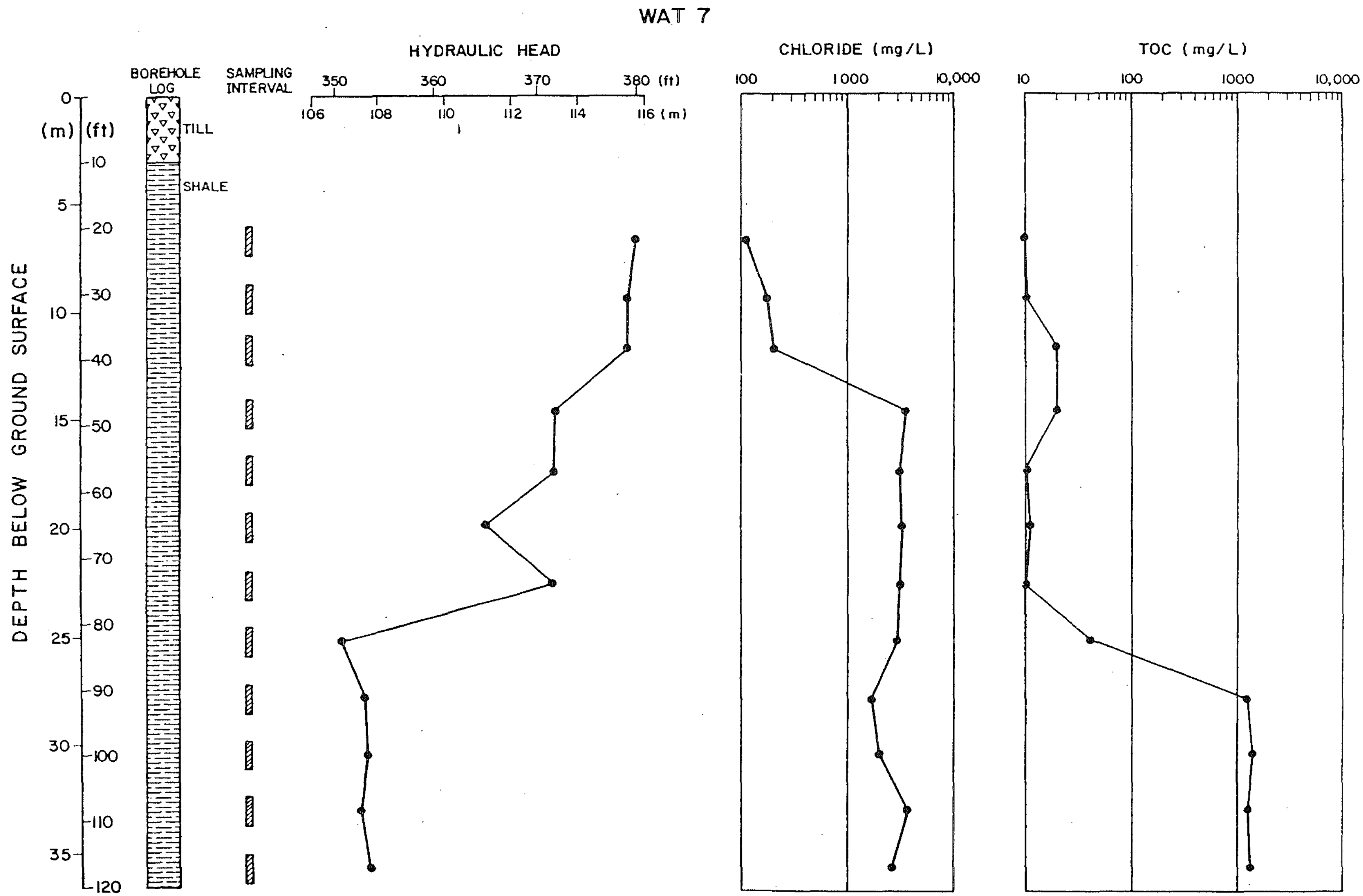

Figure 11. Depth profile of hydraulic head, chloride and total dissolved organic carbon for WAT 7.

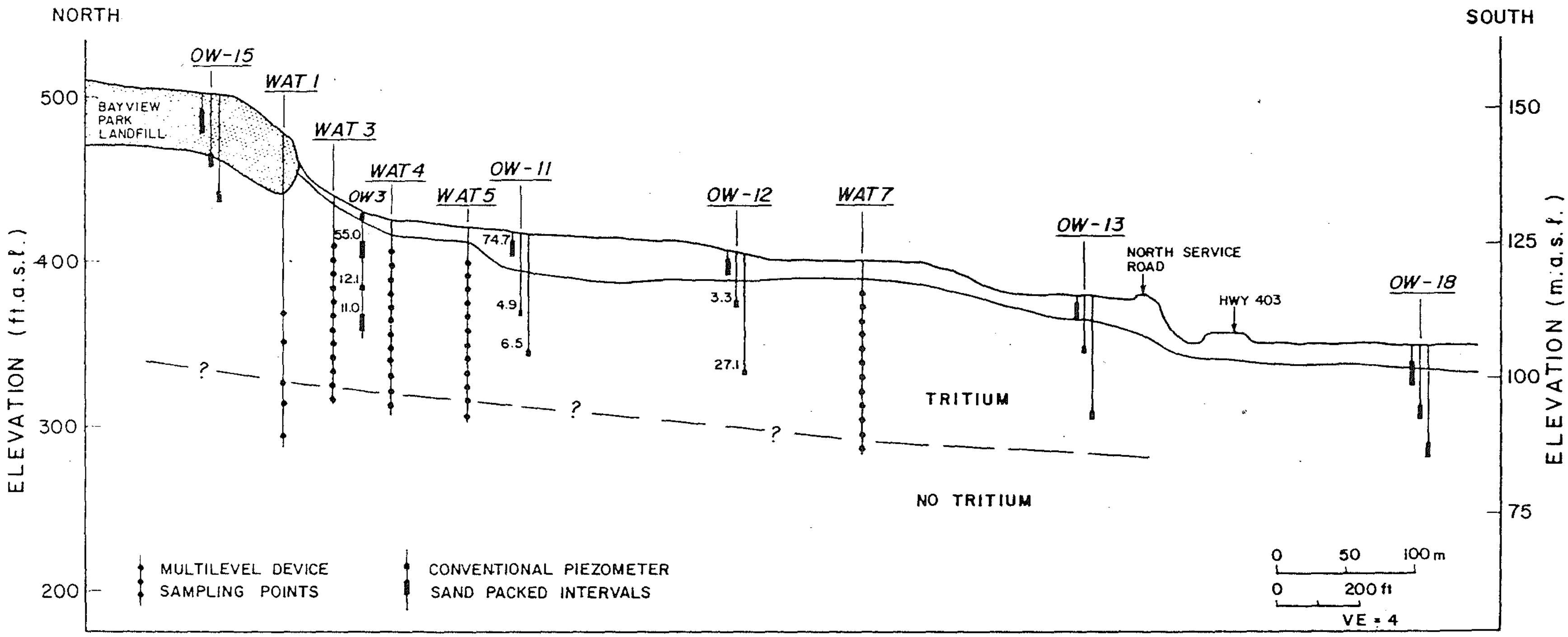

Figure 12. Tritium distribution along the main direction of groundwater flow at the Bayview Park site.