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WITNESS STATEMENT OF STANLEY FEENSTRA

In the Matter of the Consolidated Hearing for the Environmental Assessment of the Proposed Halton Landfill.

1. Introduction

I am a hydrogeochemist and President of Applied Groundwater Research Ltd. of Mississauga, Ontario. My curriculum vitae is appended to this statement.

During 1986, I was retained as a sub-consultant to Trow Hydrology Consultants Limited to provide services related to the geochemical aspects of the migration of groundwater contaminants at the candidate sites for the proposed Region of Halton Landfill. At that time, I was employed as Senior Hydrogeochemist with Zenon Environmental Inc. of Burlington, Ontario.

The results of my work were submitted to Trow Hydrology Consultants Limited in final form on July 2, 1986 in my report entitled "Evaluation of Groundwater Contaminant Migration at the Candidate Sites for the Proposed Region of Halton Landfill". This report included:

- A description of the principles of groundwater contaminant migration,
- A discussion of the chemical composition of municipal landfill leachate,
- A general discussion of the mobility of leachate constituents in groundwater,
- Evaluations of groundwater contaminant migration at Site D and Site F,
- Application of one-dimensional analytical models to simulate contaminant migration along the key groundwater pathways identified by Trow Hydrology Consultants Limited during the site investigations.

2. Landfill Leachate Chemistry

The chemical composition of the leachate which will develop in the proposed landfills at Site D or Site F would be expected to be similar to that observed in the Bayview and Burlington Landfills located adjacent to Site F. The results of chemical analyses of the Burlington and Bayview Landfill leachates are reported by Pollutech Limited (September 1986). As is typical of most municipal landfills, both leachates contain elevated concentrations of ions such as calcium, sodium, sulfate and ammonia. Metals such as iron and manganese also occur at elevated concentrations. Heavy metals such as copper, nickel, lead and zinc are detected at low concentrations. The presence of organic contaminants in the leachates is indicated by elevated concentrations of organic carbon, organic nitrogen and chemical oxygen demand (COD). The bulk of the organic compounds present in the leachates are non-toxic but still undesirable contaminants such as carboxylic acids resulting from the decomposition of organic matter in the refuse.

A variety of toxic organic chemicals have also been identified in the landfill leachates. The chemicals which are of particular interest are chlorinated and non-chlorinated volatile organic chemicals. The leachate in the Bayview Landfill was sampled and analysed for volatile organic chemicals by Pollutech Limited in January 1986 and by Conestoga-Rovers & Associates in May 1986. The leachate in the Burlington Landfill was sampled and analysed for volatile organic chemicals by Gartner Lee Limited on 11 occasions between April 1984 and May 1986 and by Pollutech Limited in October 1985. The results of the volatile organic analyses are summarized in Table 1. A wider variety of volatile organic chemicals is found in the Burlington Landfill leachate compared to the Bayview Landfill leachate. The volatile organics found in the Burlington Landfill leachate include: chlorobenzene, 1,4-dichlorobenzene, 1,1-dichloroethane, 1,2-dichloroethane, dichloromethane, tetrachloroethylene, 1,1,1-trichloroethane, trichloroethylene, ethylbenzene, toluene and xylene. The volatile organics found in the Bayview Landfill leachate include: chlorobenzene, 1,4-dichlorobenzene, ethylbenzene, toluene and xylene.

Concentrations of volatile organics in the Burlington Landfill leachate range from several hundred to several thousand µg/L (ppb) and are significantly higher than in the Bayview Landfill leachate. The Bayview Landfill operated between 1957 and 1972, and the Burlington Landfill has operated since 1972. The differences in leachate composition may be due to the difference in ages of the landfills. Laboratory leaching studies and some empirical field studies

TABLE 1. LANDFILL LEACHATE ANALYSES

Volatile Organics	Bayview Pollutech 29-Jan-86	Bayview Conestoga Rovers (1) May-86	Burlington Pollutech Oct-85	Burlington Gartner Lee Apr-84 to May-86		
				Minimum	Maximum	Mean
Chlorobenzene	46	51.9		4	50	18
1,4-Dichlorobenzene	24	18.6		10	10	10
1,1-Dichloroethane			-	11	330	131
1,2-Dichloroethane			2967	15	40	28
1,1-Dichloroethylene			4216			
Trans-1,2-dichloroethylene		1.9	-	1	3	2
Dichloromethane			390	32	2300	1069
Tetrachlorethylene			222	11	23	17
1,1,1-Trichloroethane			1400	4	236	98
Trichloroethylene	-	1.3	940	11	59	31
Trichlorofluoromethane			52			
Vinyl chloride	-	10	-			
Diethyl ether	12	-	1265			
Benzene	6.1	21.7	149	3	64	21
Ethylbenzene	26	9.4	404	9	360	86
Toluene	188	12.4	891	43	1060	470
Total Xylenes	130	74	5245	35	692	319

All concentrations reported in µg/L.

Blank results indicate concentrations less than analytical detection limit.

- Indicates chemical not reported in analysis.

(1) Maxiumum concentration detected in six leachate wells.

suggest that the concentration of many inorganic contaminants and bulk organic contaminants in landfill leachate decline with time, typically tens of years. Declines would be expected for organic chemicals but the time frame over which this reduction will take place is not known. Because volatile organic chemicals may also be removed from the landfill by volatilization and biodegradation processes in addition to leaching processes, there is some basis to believe that these chemicals may be removed from the landfill more quickly than inorganics. However, this would only be the case if no concentrated (pure chemical liquids) phases were present in the landfill.

The differences in the leachate composition between the two landfills could also be the result of differences in the nature of the waste deposited in the landfills. Because many of the volatile organics identified in the leachate are commonly used solvents, the quantities of waste containing these chemicals may have been greater with increased commercial and industrial development in area during the 1970's and 1980's. Of the two existing sites, the volatile organic chemical concentrations at Site F are expected to be more comparable to those observed at the Burlington Landfill.

3. Contaminant Mobility

For the evaluation of the proposed landfills at Site D and Site F, volatile organic chemicals are considered to be the key constituents in the landfill leachate which could potentially result in groundwater contamination. Significant concentrations of such compounds occur in the Burlington Landfill leachate at the present time and would also be expected in the leachate from Site D or Site F. These chemicals are a concern because they are generally not strongly attenuated by processes such as sorption and consequently may migrate through the subsurface at or near the rate of groundwater flow. Volatile organic chemicals in groundwater are of concern at low concentrations. Although there are no maximum acceptable concentrations (MAC) or maximum desirable concentrations (MDC) specified by the Ontario Ministry of the Environment for such chemicals, other agencies such as the US EPA and NYS DEC have specified criteria in the µg/L range for many of these chemicals in groundwater used for drinking water supply.

Based on experience at other municipal landfills and the high neutralization capability of the geologic materials at both Sites D and F, it is expected that near-neutral pH conditions will be maintained in the groundwater and toxic metals will not be mobile in the subsurface.

4. Site D

4.1 Pathways

The geological and hydrogeological conditions at Site D have been described in detail by Trow Hydrology Consultants Limited in Halton Region Landfill Technical Report Site D - Milton dated September 1986. The subsurface geological conditions at Site D consist of:

- Weathered Upper Clayey Silt Till
- Unweathered Upper Till
- Lower Silt Till (A)
- Granular Units
- Lower Silt Till (B)
- Shale Bedrock

The landfill proposed for Site D is to be excavated through the Weathered Upper Till with its base situated in the Unweathered Upper Till. A leachate collection system is to be installed on the base of the landfill to underdrain the refuse and remove leachate which develops in the landfill. The hydraulic head of leachate within the landfill will be maintained by the collection system below the water table and below potentiometric levels in all the underlying geologic units. Consequently, there will be groundwater flow into the landfill and there will be no leachate flow out of the landfill. With the leachate collection system in operation, the only potential pathway for contaminant migration out of the landfill is by diffusion downward through the Unweathered Upper Till against the relatively slow upward groundwater flow into the landfill.

In the event that the leachate collection system does not remain in operation, there will exist several potential pathways for the movement of leachate out of the landfill into the underlying geologic units. These potential pathways include:

- Lateral migration through the Weathered Upper Till,
- Downward migration through the Unweathered Upper Till and Lower Till (A) to the Granular Units with lateral migration through the Granular Units,
- Downward migration through the Upper Till, Lower Till (A & B) to the Shale Bedrock with lateral migration through the Shale Bedrock.

These were the pathways which were considered in the evaluation of groundwater contaminant migration at Site D.

4.2 Contaminant Transport Models

4.2.1 Background

The patterns of potential contaminant migration through the various geologic units at Site D were illustrated using a series of one-dimensional analytical models which simulate the effects of advection (contaminant migration due to groundwater flow), hydrodynamic dispersion, diffusion, and attenuation due to sorption on the geologic materials. The Ogata-Banks equation was used for the analytical models.

The Ogata-Banks equation is described by Bear (1979, p 268). The equation allows estimation of contaminant concentrations at various distances from a source at various times based on input values for the groundwater velocity, dispersivity and retardation factor defined by the degree of sorption of the contaminant. Because the equation uses groundwater velocity (as determined from the hydraulic conductivity, gradient and porosity) as input, its results do not differ fundamentally from simple calculations of contaminant travel time based on groundwater velocity alone. The equation does, however, allow illustration of the effect of dispersion and sorption on the rate of contaminant transport. The Ogata-Banks equation cannot take into account the effect of lateral mixing or dispersion such as occurs transverse to the direction of groundwater flow.

In the case of contaminant transport at Site D , the key pathways for contaminant migration are vertically through low hydraulic conductivity till strata and laterally through thin high hydraulic conductivity sand strata. In this situation, there is little opportunity for dispersion in directions transverse to the direction of groundwater flow, and therefore this pathway can be simulated satisfactorily using a one-dimensional model.

For the case at Site D when the *Hydraulic Trap* is in operation, the only pathway for migration of groundwater contaminants out of the landfill is downward through the Unweathered Upper Till by diffusion against the upward flow of groundwater into the landfill. A steady-state analytical solution was used to estimate the depth of penetration of contaminants by diffusion into the Unweathered Upper Till.

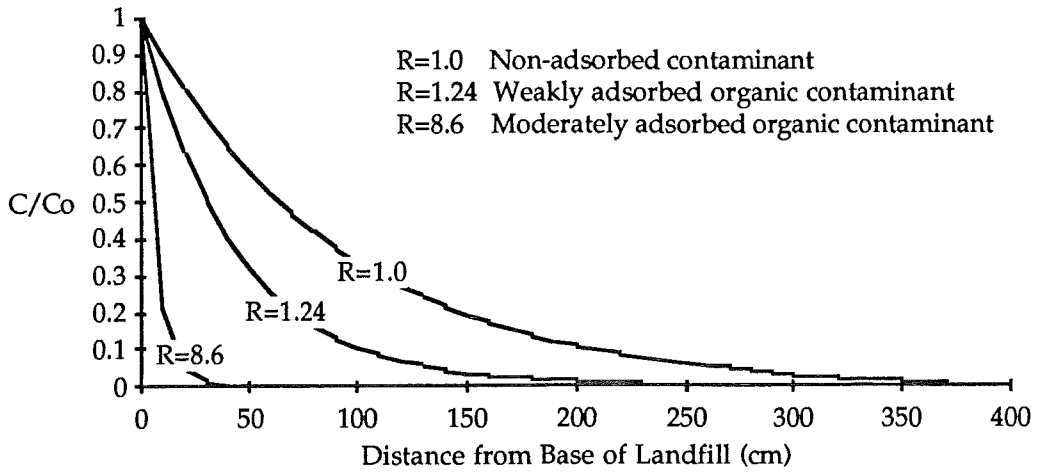
As described in my July 2, 1986 report and subsequently used in Trow Hydrology Consultants Limited report on Site D, the coefficient of molecular diffusion D^* was defined as $D^* = nD_e/R$ where D_e is the effective diffusion coefficient, n is the porosity, and R is the retardation factor. This formulation was based on my review of the current literature at the time regarding the definition of the various parameters used to describe diffusion. Since my July 2, 1986 report, I have determined that there is inconsistency in the literature and the coefficient of molecular diffusion should be defined as $D^* = D_e/R$. Because the porosity assumed for the simulation of contaminant migration in the till units was 0.25, this resulted in the use of D^* values which were too low by a factor of 4. This could potentially result in a greater degree of dispersion during contaminant migration. However, in all cases for downward contaminant migration through the till units, this change does not have a discernible effect on the results of the simulations because the advection component defined by αV_c in the formulation of the coefficient of hydrodynamic dispersion ($D = \alpha V_c + D^*$) is larger than the diffusion component defined by D^*. For the simulation of downward diffusion against upward groundwater while the *Hydraulic Trap* is in operation, the use of a higher value for the diffusion coefficient does result in deeper penetration into the Unweathered Upper Till and these revised results are reported in the following section.

4.2.2 Results

4.2.2.1 Hydraulic Trap in Operation

A revised version of the results of the case of diffusion downward into the Unweathered Upper Till against the upward flow of groundwater into the landfill during the operation of the *Hydraulic Trap* is shown in Figure 1. Reactive diffusion coefficient values used for a non-adsorbed contaminant such as chloride, a weakly adsorbed organic contaminant such as 1,2-dichloroethane and a moderately adsorbed organic contaminant such as 1,1,1-trichloroethane were 6×10^{-6} cm^2/s, 2.8×10^{-6} cm^2/s and 4.1×10^{-7} cm^2/s respectively. The diffusion coefficient value for chloride is from measured values for clay till from southwestern Ontario determined by Desaulniers (1986). Non-reactive diffusion coefficient values for the organic chemicals were estimated used the method described by Myrand (1987).

FIGURE 1. Steady state contaminant concentration in Unweathered Upper Till with downward diffusion and upward groundwater flow.



For a constant input concentration from the landfill.

The results of this simulation indicate that penetration of chloride through the Unweathered Upper Till is limited to approximately 3.0 m to 3.5 m. The penetration of weakly adsorbed organic contaminants is limited to 1.5 m to 2 m and the penetration of moderately adsorbed organics is limited to less than 0.5 m. This simulation does not take into account the additional approximately 3 m of Lower Till between the Unweathered Upper Till and the Upper Granular Unit. Although we have no information on diffusion rates through Lower Till (sandy silt till), it would not be expected to differ greatly from the Unweathered Upper Till. Upward groundwater velocities through the Lower Till and Unweathered Upper Till would be the same.

This simulation represents steady-state conditions and does not indicate what period of time is required to achieve this extent of migration. It is possible to estimate the minimum period of time required for downward diffusion by examining the rate of migration for a case in which there is no upward groundwater flow. In this case, it would require 30 to 50 years for the chloride profile shown in Figure 1 to develop. It would take considerably longer when diffusion opposes groundwater flow.

4.2.2.2 Hydraulic Trap Not in Operation

The simulation of contaminant migration laterally through the Weathered Upper Till indicate that the first arrival (C/C_O=0.1) of a non-adsorbed contaminant (chloride) would occur at a distance of 100 m in approximately 5 years (see Trow Hydrology Consultants Limited Site D Report, Figure 16, Page 112). The migration of adsorbed contaminants was not simulated in this case because it is believed that groundwater flow in this zone will occur primarily through fissures in the till and the sorptive properties of the fissure surfaces cannot be readily estimated. Similarly, the effect of matrix diffusion was not accounted for in the simulation because it was considered that specific knowledge of the physical and hydraulic properties of the fissures was not sufficient to justify the application of relatively complex analytical or numerical models which can simulate matrix diffusion. Both sorption and matrix diffusion will act to attenuate contaminant migration resulting in longer first arrival times and lower peak concentrations at the site boundary.

The relatively rapid contaminant migration rate predicted for the Weathered Upper Till is a result of the the relatively high hydraulic conductivity used and the low porosity value estimated for the weathered till. Theoretically, if groundwater flow in the weathered till occurs through the fissures, given the hydraulic conductivity of the weathered till and the estimated fissure frequency, the transport porosity (ie. the porosity which actually transmits water) should be in the range of 0.001 to 0.00001. The value of 0.0001 cited by Gartner Lee Limited in their report to OWMC on the Site D area was attributed to Desaulniers *et al* . (1981). However, this paper deals with the occurrence and nature of groundwater in unweathered clay tills and I could find no reference to this fissure porosity value in the paper by Desaulniers. I believe that the 0.0001 porosity value is derived from a field tracer test and a theoretical analysis performed by Grisak (1975) in fissured tills in southern Manitoba. It is my understanding that the fissured glacial tills in western Canada differ considerably from clayey till in southern Ontario (personal communication: Dr. John Cherry) and it is unlikely that this value can be usefully applied to Ontario conditions.

Experience in Ontario is such that extensive and rapid migration of dissolved groundwater contaminants through weathered till, which would have to occur if porosity are as low as 0.0001, is not observed at waste disposal facilities. However, it is important to recognize that although the transport porosity may be in the range of 0.0001 and the actual movement of water through the fractures will be rapid, contaminant migration through the fractures will be

attenuated due to the effect of matrix diffusion. Therefore, in terms of calculation of contaminant migration through fractured till there will also be an apparent transport porosity for the contaminants which will be higher (ie. slower velocity) than the transport porosity for the water.

In selection of an appropriate porosity value to apply to the Weathered Upper Till at Site D, the study team was confronted with the very low value from theoretical calculations and a single test in western tills, observations in southern Ontario which indicated much slower contaminant migration rates and therefore a much higher apparent porosity. Given these extremes, a relatively low porosity value of 0.001 was selected to err on the side which would over-estimate contaminant migration rates through the Weathered Upper Till. Since the Site D study in 1986, studies have been recently completed by the University of Waterloo in the Sarnia area which included a bromide tracer test to examine contaminant migration between two trenches in the weathered zone (personal communication: Dr. John Cherry) . Although the analysis of the data from this study has not been completed at this time, the travel time through the 2 m of till separating the trenches was 3 weeks with a high hydraulic gradients (~0.4). This rate of migration yields an apparent transport porosity in the range of 0.02 considerably higher than the transport porosity for water of 0.0002 to 0.0008 calculated on the basis of the hydraulic conductivity and fracture spacing. This apparent transport porosity is higher than that 0.001 used by Trow Hydrology Consultants Limited for the weathered till at Site D, suggests that the value used by Trow Hydrology Consultants Limited at Site D is conservative in that it would result in higher contaminant velocities than which may be observed.

With a declining input concentration from leachate at the landfill, the simulations indicate first arrival times (C/Co=0.1) of a non-adsorbed contaminant (chloride) and a weakly adsorbed contaminant (1,2-dichloroethane) at the site boundary to be approximately 70 and 90 years respectively. The peak concentrations experienced at the site boundary in the granular unit are lower than the peak landfill concentrations by a factor of approximately 5 times (see Trow Hydrology Consultants Site D Report, Figures 19, 20 and 21, Pages 114, 115 and 116).

The pathway for migration downard through the till units to the shale bedrock was considered to be of minor significance because travel times through the lower portion of the Lower Till would be measured in hundreds of years (see Trow Hydrology Consultants Site D Report, Figures 12, Page 103).

4.3 Conclusions

For conditions where the *Hydraulic Trap* is in operation, there should be no migration of contaminants from the landfill into the groundwater within the underlying granular units. For conditions where the *Hydraulic Trap* is not in operation, the migration of contaminants will occur downward through the Unweathered Upper Till, and Lower Till and into the Granular Units. Within the Granular Units, contaminants will migrate laterally off-site. Leakage of leachate from the landfill will eventually result in elevated concentrations of contaminants in the groundwater in the Granular Units. Contaminant migration simulations indicate chloride concentrations in the Granular Units could be increased by approximately 300 to 400 mg/L. These concentrations would be in addition to the several hundred mg/L chloride which would be expected naturally in the groundwater in the Granular Units, and would consequently result in chloride concentrations in excess of the MOE Maxiumum Desirable Concentration of 250 mg/L. It is estimated that this impact could occur in an estimated several tens of years following failure of the *Hydraulic Trap* .

5. Site F

5.1 Pathways

The geological and hydrogeological conditions at Site F have been described in detail by Trow Hydrology Consultants Limited in Halton Region Landfill Technical Report Site F - Burlington dated September 1986. The subsurface geological conditions at Site F consist of:

- Weathered Moderately Fractured Shale
- Weakly Fractured Shale
- Highly Fractured Shale
- Moderately Fractured Shale

The landfill proposed for Site F will be underlain by Weathered Moderately Fractured Shale. Lateral migration of contaminants through the Weathered Shale will occur but this discharge

will be contained by the perimeter drain around the south side of the landfill and was not considered as a pathway for off-site migration of groundwater contaminants. The Highly Fractured Shale underdrains the overlying strata and will be the dominant pathway for the migration of groundwater contaminants off-site through the deeper shale at Site F. The Highly Fractured Shale zone is interpreted to subcrop approximately 1400 m south of the landfill where it is overlain by a clayey silt till which is referred to as the Discharge Zone Till. The pathway downward through the Weakly Fractured Shale, laterally through the Highly Fractured Shale and upward through the Discharge Zone Till is the key route for off-site migration of groundwater contaminants from the landfill.

5.2 Background Groundwater Chemistry

Previous investigations have determined that the groundwater in the shale bedrock underlying Site F contains naturally elevated concentrations of most of the inorganic and organic constituents which are typically used by hydrogeologists as chemical indicators of groundwater contamination by landfill leachates. Naturally occurring concentrations of chloride, sulfate, sodium, hardness, alkalinity, dissolved organic carbon, and phenols in the groundwater in the shale are variable and frequently higher than those in landfill leachate and consequently cannot be used to indicate contamination from a landfill.

The use of organic nitrogen concentrations as an indicator of landfill leachate contamination in groundwater was suggested by Conestoga-Rovers & Associates in their 1986 Monitoring Report for the Bayview Landfill Site. Conestoga-Rovers & Associates determined the background (natural) organic nitrogen concentrations in groundwater in the shale to be less than 0.3 mg/L based on the results of only two monitoring wells. The mean organic nitrogen concentration in the leachate at the Bayview landfill was 23 mg/L. Therefore, Conestoga-Rovers & Associates concluded that organic nitrogen concentrations greater than 0.3 mg/L in the groundwater should represent contamination by landfill leachate south of the Bayview landfill. However, little is known about the occurrence of organic nitrogen in groundwater or the natural variability of organic nitrogen concentrations. The interpretation of 0.3 mg/L organic nitrogen as background for the Site F shale is tenuous in light of concentrations ranging from 0.35 to 1.0 mg/L in GLA-23 located upgradient of the Burlington Landfill, and background concentrations of approximately 1 mg/L in sand aquifers at the CFB Borden Landfill (Nicholson *et al.*, 1983) and the North Bay Landfill (Buska, 1980).

5.3 Contaminant Migration at Bayview and Burlington Landfill Sites.

5.3.1 Observations

Burlington Landfill

Annual monitoring of the chemical composition of the groundwater down-gradient of the Burlington Landfill has been conducted by Gartner Lee Limited since 1980. It has not been possible to delineate any groundwater contamination resulting from the landfill on the basis of inorganic or routine organic water quality parameters because of the naturally occurring variable and high concentrations of these parameters in the groundwater in the shale bedrock. During November and December 1986, groundwater samples were collected from 12 monitoring wells around the Burlington Landfill on two or three occasions and were analysed for volatile organic chemicals. The groundwater samples were analysed for Gartner Lee by the Ontario Ministry of the Environment Laboratories.

Only trace concentrations of trichloroethylene, tetrachloroethylene and 1,1,1-trichloroethane were detected in several of the monitoring wells. These results are shown below:

Compound	Location	Well	Concentration (µg/L)
1,1,1-Trichloroethane	Up-gradient of landfill	GLA-23-I	1 - 2
	Up-gradient of landfill	GLA-23-II	<1 - 5
	25 m down-gradient	GLA-24-III	<1 - 3
	Near Hwy 403	GLA-26-II	<1 - 1
	Near Hwy 403	GLA-27-I	<1 - 2
Trichloroethylene	25 m down-gradient	GLA-24-III	<1 - 9
Tetrachloroethylene	25 m down-gradient	GLA-24-III	<1 - 3

The concentrations of 1,1,1-trichloroethane are low and sporadic and Gartner Lee concluded that these concentrations were not related to contamination from the landfill. Gartner Lee Limited attributed the low concentrations of trichloroethylene and tetrachloroethylene in GLA-24-III to be a result of the migration of leachate from the landfill, and reasoned that the concentrations are much lower than in the leachate due to attenuation processes such as sorption, biodegradation and matrix diffusion which will act to reduce the concentrations in the groundwater. Based on this finding, Gartner Lee concludes that groundwater contamination from

the landfill extends to only a shallow depth and a distance of approximately 25 m from the landfill.

Although I would agree concentrated leachate containing high concentrations of volatile organic chemicals does not occur in any of the monitoring wells at the Burlington site, the storage time for the Gartner Lee samples may have resulted in a loss of volatiles prior to analysis. This means that the actual concentrations of volatile organics in the groundwater samples may be detectable in some samples which were determined to be less than detection and higher in samples which were determined to have low concentrations. However, I would conclude that the concentrations of volatile organics, if present in the groundwater, will be substantially lower than those determined in the leachate indicating a significant degree of attenuation during migration through the shale bedrock.

Bayview Landfill

The migration of groundwater contaminants from the Bayview Landfill was investigated by the University of Waterloo (UW) during 1983. Groundwater samples were collected from a variety of conventional piezometers and multilevel piezometers installed to the south of the Bayview Landfill. The key findings of the study by the UW were reported by Hewetson (1985) and describe the occurrence of 1,1,1-trichloroethane, chlorobenzene and 1,4-dichlorobenzene in the groundwater south of the Bayview Landfill. During this study some additional information on the concentrations of benzene, toluene, ethylbenzene, trichloroethylene and tetrachloroethylene in the groundwater samples was produced and was not reported. I received this information on September 17, 1987 from Dr. James Pankow at the Oregon Graduate Centre, who had performed the chemical analyses for the UW study. A complete listing of the groundwater analyses of 35 piezometers sampled by UW for volatile organic chemicals is included in Table 2. These data have not be reported elsewhere. The results of the UW groundwater analyses are summarized in the following:

TABLE 2. RESULTS OF UW VOLATILE ORGANIC ANALYSES FOR BAYVIEW GROUNDWATERS

Sample	Well	1,1,1-TCA	Benzene	TCE	Toluene	PCE	Chlorobenzene	Ethylbenzene	1,4-DCB	1,2-DCB
459	BLANK	0.0333	0.109	*ND*	0.05	0.0123	0.00352	0.0035	*ND*	0.00028
466	BLANK	0.219	0.0872	0.00204	0.158	0.0217	0.0248	0.0223	0.00177	0.00436
473	BLANK	0.0618	0.115	0.00418	0.149	0.0258	0.00964	0.0219	0.00119	0.00398
478	BLANK	0.175	0.136	0.00558	0.112	0.0132	0.0237	0.0452	0.0011	0.0115
493	BLANK	0.0507					0.00368		0.000491	0.00185
429	BLANK	0.019	0.156	*ND*	0.0292	0.00837	0.00425	0.00152	0.000386	0.00151
435	BLANK	0.0774	0.016	*ND*	0.0548	0.031	*ND*	0.00873	*ND*	0.00188
689	BLANK	0.0266	0.0937	*ND*	0.0432	0.021	*ND*	0.00526	*ND*	*ND*
401	BLANK	*ND*	0.0586	*ND*	0.0888	0.0181	*ND*	0.0099	*ND*	*ND*
496	BLANK	0.00538	0.0119	*ND*	0.0221	0.0169	*ND*	*ND*	*ND*	*ND*
412	BLANK	0.0164	0.0437	*ND*	0.027	0.0547	*ND*	0.00294	*ND*	*ND*
421	BLANK	*ND*	0.0103	*ND*	0.0313	*ND*	*ND*	*ND*	*ND*	*ND*
418	BLANK	*ND*	0.0157	*ND*	0.029	*ND*	0.00299	*ND*	*ND*	*ND*
441	BLANK	0.321	0.0169	*ND*	0.0465	0.726	*ND*	0.0035	*ND*	*ND*
448	BLANK	0.0231	0.0135	*ND*	0.0326	0.0185	*ND*	0.00301	*ND*	*ND*
670	BLANK	0.0113	0.0201	0.00129	0.0609	0.0167	*ND*	0.00468	*ND*	*ND*
443	BLANK	0.0229	0.0189	*ND*	0.0813	0.0536	*ND*	0.00515	*ND*	*ND*
435	BLANK	*ND*	0.0256	*ND*	0.0733	0.0346	*ND*	0.00709	*ND*	*ND*
433	BLANK	0.00634	0.0129	*ND*	0.0485	0.0419	*ND*	0.00587	*ND*	*ND*
561	BLANK	*ND*	0.077	*ND*	0.0703	0.024	*ND*	*ND*	*ND*	*ND*
	Maximum	0.321	0.156	0.00558	0.158	0.726	0.0248	0.0452	0.00177	0.0115
494	MB-1-2	**3.92**	**0.567**	**0.184**	**1.13**	*LTB*	**13.6**	*LTB*	**1.53**	**0.133**
687	MB-1-2	**3.47**	**0.646**	**0.223**	**1.01**	*LTB*	**13.2**	*LTB*	**1.57**	**0.135**
499	MB-1-4	**20.5**	*LTB*	**0.0119**	**5.81**	*LTB*	*ND*		*LTB*	*LTB*
683	MB-1-4	**15.4**	*LTB*	*ND*		*LTB*	*LTB*	**0.0899**	*LTB*	*ND*
684	MB-1-4	**22.5**	**0.16**	**0.0234**	**6.79**	*LTB*	*LTB*	**0.0747**	*LTB*	*LTB*
431	MB-11-1	*LTB*	**0.222**	**0.00779**	*LTB*	*LTB*	**3.38**	*LTB*	**0.566**	**0.0705**
432	MB-11-1	*LTB*	**0.253**	**0.0129**	*LTB*	*LTB*	**3.35**	*LTB*	**0.583**	**0.0696**
437	MB-11-2	**2.63**	*LTB*	*LTB*	*LTB*	*LTB*	*ND*	*LTB*	*ND*	*LTB*
438	MB-11-2	**2.44**	*LTB*	*LTB*	*LTB*	*LTB*	*LTB*	*LTB*	*ND*	*ND*
491	MB-11-3	**6.85**	**0.189**	**0.0174**	**0.327**	*LTB*	**0.0336**	**1.02**	**0.00263**	*ND*
492	MB-11-3	**5.18**	*LTB*	**0.0173**	**0.335**	*LTB*	**0.0256**	**0.819**	**0.00427**	*LTB*
427	MB-12-2	**7.06**	*LTB*	**0.0425**	*LTB*	*LTB*	LTB	*LTB*	*LTB*	*LTB*
428	MB-12-2	**7.24**	**0.165**	**0.0365**	*LTB*	*LTB*	LTB	*LTB*	*LTB*	*LTB*
430	MB-12-3	**4.67**	*LTB*	**0.0156**	**1.34**	*LTB*	**0.602**	**0.0919**	**0.0559**	**0.0116**
434	MB-12-3	**4.19**	*LTB*	**0.0189**	**1.17**	*LTB*	**0.58**	**0.1**	**0.051**	**0.0142**
483	MB-13-2	**12.6**							*LTB*	*LTB*
484	MB-13-2	**11.4**					**0.00929**		*LTB*	*LTB*

All concentrations in µg/L.
ND - Not detected at nominal detection limit 0.0001 µg/L. LTB - Less than maximum blank concentration.

TABLE 2. RESULTS OF UW VOLATILE ORGANIC ANALYSES FOR BAYVIEW GROUNDWATERS

Sample	Well	1,1,1-TCA	Benzene	TCE	Toluene	PCE	Chlorobenzene	Ethylbenzene	1,4-DCB	1,2-DCB
425	MB-13-3	4.28							0.006	*LTB*
426	MB-13-3	4.28	*LTB*	0.025	1.19	*LTB*	*LTB*	0.202	0.00636	*LTB*
455	MB-14	4.41	*LTB*	0.0107	0.499	*LTB*	*ND*	*LTB*	0.00812	*LTB*
456	MB-14	3.4	*LTB*	0.00676	0.347	*LTB*	*LTB*	*LTB*	0.0054	*LTB*
415	MB-15-3	9.72	*LTB*	0.0989	5.96	*ND*		0.256	0.222	*ND*
414	MB-15-3	12.5	0.259	0.178	6.95	3.74	2.61	0.356	0.216	*ND*
417	MB-15-3	13.1	0.202	0.114	6.18	*LTB*	2.23	0.35	0.27	*LTB*
474	MB-16-1	*LTB*	*LTB*	*LTB*	1.6	*LTB*	*ND*	1.24	0.00258	*LTB*
477	MB-16-1	*LTB*	0.166	*LTB*	1.87	*LTB*	*ND*	1.29	0.00284	*LTB*
475	MB-16-2	*LTB*	*LTB*	*LTB*	5.18	*LTB*	*LTB*	1.18	0.00344	*LTB*
476	MB-16-2	*LTB*	*LTB*	*LTB*	2.21	*LTB*	*LTB*	1.1	0.00355	0.0117
479	MB-16-3	3.66							0.0692	0.0546
481	MB-16-3	3.25							0.0619	0.0593
467	MB-17-2	0.707	0.715	*ND*	75.2	*LTB*	*ND*	3.24	0.0489	*ND*
468	MB-17-2	0.763	2.83	0.00937	133	*LTB*	*LTB*	4.4	0.0499	0.532
469	MB-17-3	*LTB*	0.436	*LTB*	2.43	*LTB*	*LTB*	*LTB*	*ND*	*LTB*
471	MB-17-3	*LTB*	*LTB*	*LTB*	0.902	*LTB*	*LTB*	*LTB*	*LTB*	*LTB*
461	MB-18-1	0.589	*LTB*	*LTB*	6.98	*LTB*	*ND*	1.12	0.00856	*LTB*
465	MB-18-1	0.511	*LTB*	*LTB*	7.01	*LTB*	*ND*	1.07	0.00848	*LTB*
464	MB-18-2	*LTB*	0.174	*LTB*	21	*LTB*	*ND*	0.345	0.0142	*LTB*
463	MB-18-2	*LTB*	0.192	*LTB*	23.5	*LTB*	*ND*	0.362	0.0131	*LTB*
457	MB-5	*LTB*	*LTB*	0.00664	0.435	*LTB*	*ND*	*LTB*	0.00855	*LTB*
458	MB-5	*LTB*	*LTB*	*LTB*	0.26	*LTB*	*ND*	*LTB*	*ND*	*LTB*
685	WAT-2-3	5.13	0.347	*ND*	4.31	0.745	*ND*	0.136	0.232	*ND*
402	WAT-2-3	5.29	0.387	0.0233	4.14	*LTB*	*ND*	0.132	0.207	*ND*
408	WAT-2-7	3.07	0.227	0.0185	2.91	*LTB*	*ND*	*LTB*	0.179	*ND*
409	WAT-2-7	3.14	0.247	0.0207	2.76	*LTB*	*ND*	0.0804	0.183	*ND*
563	WAT-3-11	0.747	*LTB*	0.232	0.269	*LTB*	3.68	*LTB*	0.318	0.0207
529	WAT-3-11	0.683	0.162	0.285	0.271	*LTB*	3.07	*LTB*	0.274	0.0153
449	WAT-3-2	1.49	0.323	0.55	0.836	*ND*	4.07	0.0459	0.269	*ND*
451	WAT-3-2	2.12	0.849	0.768	3.33	0.784	5.18	0.113	0.369	*ND*
504	WAT-3-7	1.23	0.291	0.163	0.294	*LTB*	1.75	*LTB*	0.163	*ND*
558	WAT-3-7	1.01	*LTB*	0.226	0.306	*LTB*	1.79	*LTB*	0.146	*LTB*
442	WAT-3-7	0.983	0.174	0.194	0.257	*LTB*	1.2	*LTB*	0.0946	*LTB*
406	WAT-4-12	7.53	5.98	0.116	13	*LTB*	1.43	3.69	0.24	0.0129
407	WAT-4-12	7.92	7.2	0.142	15.9	*LTB*	1.48		0.192	0.0105
538	WAT-4-2	3.06	5.31	0.0438	9.73	*ND*	0.592	3.01	0.173	*ND*
576	WAT-4-2	5.02	8.96	0.1	23.4	*LTB*	0.629	7.02	0.253	0.0158
889	WAT-4-5	8.83	9.02	*ND*	29.7	*ND*		4.24	0.147	*ND*

All concentrations in µg/L.
ND - Not detected at nominal detection limit 0.0001 µg/L. LTB - Less than maximum blank concentration.

TABLE 2. RESULTS OF UW VOLATILE ORGANIC ANALYSES FOR BAYVIEW GROUNDWATERS

Sample	Well	1,1,1-TCA	Benzene	TCE	Toluene	PCE	Chlorobenzene	Ethylbenzene	1,4-DCB	1,2-DCB
403	WAT-4-5	10.2	18.1	*ND*	40.3	*LTB*	0.817	7.71	0.157	0.00925
404	WAT-4-8	9.55	13.1	*ND*	25.6	*LTB*	0.661	3.27	0.0819	*ND*
405	WAT-4-8	9.04	24.3	*ND*	23	*LTB*	0.832	4.37		*ND*
444	WAT-5-10	1.68	0.35	0.061	3.38	*LTB*	1.78	*LTB*	0.0408	*ND*
445	WAT-5-10	1.82	0.353	0.059	3.37	*LTB*	1.93	*LTB*	0.0477	*ND*
446	WAT-5-12	2.27	0.575	0.089	3.87	*ND*	2.84	0.112	0.126	0.0156
447	WAT-5-12	1.94	0.494	0.0904	3.37	*LTB*	2.81	0.159		
423	WAT-5-3	1.15	0.453	0.0563	2.73	*ND*	2.55	0.0674	0.134	*ND*
424	WAT-5-3	1.52	0.337	0.0503	2.8	*LTB*	2.2	0.058	0.117	0.0128
439	WAT-5-8	1.74	0.369	0.062	3.45	*LTB*	3.06	0.0786	0.144	0.0165
440	WAT-5-8	1.87	0.328	0.0672	4.12	*LTB*	2.5	0.0564	0.116	*ND*
420	WAT-7-11	28.7	*LTB*	*ND*	12.7	*LTB*	*LTB*	0.0956	*LTB*	*ND*
422	WAT-7-11	27.5	0.171		9.94	*LTB*	*ND*	0.0901	*LTB*	*ND*
410	WAT-7-5	8.55	*LTB*	0.0399	7.2	*LTB*	*LTB*	0.0512	0.0191	*ND*
411	WAT-7-5	9.74	*LTB*	0.0298	6.27	*LTB*	*ND*	0.059	0.023	*ND*
436	WAT-7-9	29.1	0.277	0.0381	12.2	*ND*	*ND*	0.0788	0.0173	0.0493
419	WAT-7-9	26.7	0.218	*ND*	12.2	*LTB*	*ND*	*ND*	*ND*	*ND*

All concentrations in µg/L.
ND - Not detected at nominal detection limit 0.0001 µg/L. LTB - Less than maximum blank concentration.

Chemical	Minimum	Maximum	Geometric Mean	Drinking Water Standard	
1,1,1-Trichloroethane	ND	29.1	2.1	200 µg/L	US EPA
Chlorobenzene	ND	13.6	0.024	20 µg/L	NYS DEC
1,4-Dichlorobenzene	ND	1.57	0.023	75 µg/L	US EPA
Benzene	ND	24.3	0.28	5 µg/L	US EPA
Toluene	0.056	133	2.5	50 µg/L	NYS DEC
Ethylbenzene	ND	7.71	0.12	50 µg/L	NYS DEC
Trichloroethylene	ND	0.768	0.008	5 µg/L	US EPA
Tetrachloroethylene	ND	3.74	0.047	2 µg/L	NYS DEC
1,2-Dichlorobenzene	ND	0.532	0.002	4.7 µg/L	NYS DEC

All concentrations in µg/L. ND - Not detected at approximately 0.0001 µg/L.
US EPA - United States Environmental Protection Agency
NYS DEC- New York State Department of Environmental Conservation

One or more of the above listed volatile organic chemicals were detected in nearly all the groundwater samples collected by the UW study. Volatile organic chemicals were detected in the groundwater at depths of 40 m into the shale (the deepest piezometers sampled) and at a distance of up to 900 m from the landfill (the farthest piezometer sampled).

The UW study concluded that 1,1,1-trichloroethane, chlorobenzene and 1,4-dichlorobenzene were contaminants derived from the landfill. Benzene, toluene and ethylbenzene were considered to have been derived from naturally occurring petroleum compounds in the shale bedrock. Trichloroethylene, tetrachloroethylene and 1,2-dichlorobenzene occurred at very low levels and were common in the sample cartridge blanks so that these chemicals were not considered to represent chemical concentrations in the samples.

As can be seen from Table 2 and the summary table above, none of the groundwater samples contained concentrations of 1,1,1-trichloroethane, chlorobenzene, 1,4-dichlorobenzene, ethylbenzene, trichloroethylene, or 1,2-dichlorobenzene which exceeded US EPA or NYS DEC drinking water standards. Only a small number of samples exceeded standards for benzene and toluene and only a single sample exceeded the standard for tetrachloroethylene.

Since completion of the UW sampling and analysis at the Bayview site in early 1984, the state-of-the-art regarding well installation and sampling procedures for organic chemicals in groundwater has advanced considerably. Also additional hydrogeological studies, groundwater analyses and leachate analyses have been completed at Site F, and at the Bayview and Burlington Landfills. Based on the results of the UW study and these additional studies, it is my opinion that there is sufficient evidence to suggest that the chemical concentrations determined in the UW study do not reflect actual groundwater concentrations and likely result from contamination derived from the piezometer installations. The following points summarize the relevant evidence:

- The Waterloo Multilevel piezometers (WAT wells) and Morrison Beatty piezometers (MB wells) which were sampled in the UW study were constructed of PVC pipe with solvent welded joints and the pipe was not cleaned prior to use for the installations. At the time of the UW and MB installations it was not common practise to pre-clean pipe and avoid solvent welded joints. Both solvent welded joints and uncleaned pipe can contribute to contamination of the groundwater samples collected from the installations. The effects of this type of contamination are likely exacerbated by the relatively low hydraulic conductivity of the shale and the generally small volumes of groundwater purged from the installations before sampling.

- Leakages which interconnected the water inside the casing with the groundwater in the sampling interval were noted following installation of the Waterloo Multilevel piezometers (see Hewetson, 1985, p. 7). Although leakages were corrected by grouting the inside of the casings, the small volumes of water purged from the sampling intervals (maximum 6 L) before sampling would not likely have been sufficient to completely remove the leaked water.

- The benzene, toluene and ethylbenzene concentrations may represent chemicals which occur naturally in the shale bedrock but none of these compounds have been detected (<1 µg/L) in the GLA wells at locations 23, 24, 26 and 27 at the Burlington Landfill or HC-15. All of these wells are of stainless steel construction with threaded joints and were pre-cleaned before

installation. Likewise no benzene, toluene or ethylbenzene was detected (<1 µg/L) in the wells at location HC-9 which are PVC wells with threaded joints. HC-9 is immediately adjacent to MB-18-1 and MB-18-2 in which toluene was detected 7 µg/L and 21.8 µg/L respectively. Other chemicals were also detected in MB-18 by UW but at lower concentrations than could be detected by the laboratories used by Gartner Lee Limited and Trow Hydrology Consultants.

- Various of the volatile organic chemicals detected in the groundwater down-gradient of the landfill were also detected in the background piezometers MB-5, MB-14, WAT-2-3 and WAT-2-7.

Chemical	MB-5	MB-14	WAT-2-3	WAT-2-7
1,1,1-Trichloroethane	LTB	3.9	5.2	3.1
Chlorobenzene	ND	LTB	ND	ND
1,4-Dichlorobenzene	0.0004	0.0068	0.22	0.18
Benzene	LTB	LTB	0.37	0.24
Toluene	0.35	0.42	4.2	2.8
Ethylbenzene	LTB	LTB	0.13	0.06
Trichloroethylene	0.006	0.008	0.01	0.039
Tetrachloroethylene	LTB	LTB	0.38	LTB
1,2-Dichlorobenzene	LTB	LTB	ND	ND

All concentrations in µg/L. ND - Not detected at approximately 0.0001 µg/L. LTB - Less than maximum reported concentration on blank cartridges.

- No 1,1,1-trichloroethane or common degradation products of 1,1,1-trichloroethane such as 1,1-dichloroethane and chloroethane were detected (<1 µg/L) by Conestoga-Rovers & Associates in any of the six leachate monitoring wells within the Bayview Landfill.

- If 1,1,1-trichloroethane were present in the deeper groundwater in the shale, concentrations of common degradation products such as 1,1-dichloroethane would also be expected but none were detected in the UW study.

Based on this evidence it is my opinion, that most or all of the volatile organic chemical concentrations detected in the UW study do not reflect actual groundwater concentrations and are likely the result of extraneous contamination from the piezometer installations.

3.3.2 Implications for Site F

The results of the investigations at the Burlington and Bayview Landfills suggest that operation of unengineered and partially engineered municipal landfills in this hydrogeologic setting over periods of up to 30 years has not resulted in significant contamination of the groundwater beneath or down-gradient of the sites.

3.4 Contaminant Transport Models

3.4.1 Background

The one-dimensional formulation of contaminant migration (Ogata-Banks equation) was considered to be satisfactory for use at Site F because the key pathway for contaminant migration is vertically through low hydraulic conductivity Weakly Fractured Shale and laterally through relatively thin high hydraulic conductivity highly fractured shale zones. In this situation, there is little opportunity for dispersion in directions transverse to the direction of groundwater flow.

As I describe earlier in this witness statement, in my July 2, 1986 report, the coefficient of molecular diffusion D^* was defined as $D^* = nD_e/R$ where D_e is the effective diffusion coefficient, n is the porosity , and R is the retardation factor. This formulation was based on the current literature at the time regarding the definition of the various parameters used to describe diffusion. Since my July 2, 1986 report, I have determined that there is inconsistency in the literature and the coefficient of molecular diffusion should be defined as $D^* = D_e/R$. Because the porosity assumed for the simulation of contaminant migration in the Discharge Zone Till unit was 0.25, this resulted in the use of D^* values which were too low by a factor of 4 times . This could potentially result in a greater degree of dispersion during contaminant migration. However, in all cases for downward contaminant migration through the till unit, this change does not have a discernible effect on the results of the simulations because the advection component defined by αV_c in the formulation of the coefficient of hydrodynamic dispersion ($D = \alpha V_c + D^*$) is larger than the diffusion component defined by D^* .

3.4.2 Results

Simulations of contaminant migration were conducted for the Weakly Fractured Shale - Highly Fractured Shale - Discharge Zone Till pathway. For a declining input concentration from leachate at the landfill, the simulations indicate first arrival times at the discharge zone of a non-adsorbed contaminant (chloride) and a weakly adsorbed contaminant (1,2-dichloroethane) at the site boundary to be approximately 60 and 100 years respectively. The peak concentrations experienced at the discharge zone in the granular unit are lower than the peak landfill concentrations by a factor of approximately 3 times. The results of these simulations are shown are shown in the Trow Hydrology Consultants report on Site F (Figures 18, 19 and 20 on pages 119, 120 and 121 respectively).

The groundwater velocity through the Highly Fractured Shale is relatively high due to the high hydraulic conductivity determined for the zone from the pumping test in PW-1 and the relatively low porosity value used to reflect groundwater flow primarily through the fractures. For the individual component model for simulation of contaminant migration through the Highly Fractured Shale, the first arrival of contaminants at the discharge zone till would be less than a year. Because the very rapid velocity predicted for the Highly Fractured Shale was much greater than the velocities through the Weakly Fractured Shale or Discharge Zone Till, this component model was not coupled specifically into the pathway model.

The migration of adsorbed contaminants was not simulated in the Highly Fractured Shale case because it is believed that groundwater flow in this zone will occur primarily through fractures and the sorptive properties of the fracture surfaces cannot be readily estimated. Similarly, the effect of matrix diffusion was not accounted for in the simulation because it was considered that specific knowledge of the physical and hydraulic properties of the fractures was not sufficient to justify the application of relatively complex analytical or numerical models which can simulate matrix diffusion. Both sorption and matrix diffusion will act to significantly attenuate contaminant migration, resulting in longer first arrival times and lower peak concentrations at the discharge zone.

3.5 Conclusions

The results of the simulations of contaminant migration suggest that eventual off-site migration of groundwater contaminants will occur as a result of landfill operations at Site F. These impacts would be evident at the discharge zone in approximately 60 years (see Trow Hydrology

Consultants report on Site F, Figures 19, 20 and 21 on pages 119, 120 and 121 respectively) and would be evident in the shale bedrock beneath the site in approximately 30 years (seeTrow Hydrology Consultants report on Site F, Figures 13a and 13b on pages 107 and 108). This prediction is generally consistent with the observation that there is currently no significant contamination of deeper groundwaters beneath the Bayview Landfill which is 30 years old, and the Burlington Landfill which is 15 years old. The contaminant migration simulations could not take into account the effects of matrix diffusion or biodegradation. Attenuation due to these processes will serve to reduce concentrations and lengthen travel times through the subsurface, and minimize whatever environmental impact which could result from leakage of leachate from the landfill at Site F.

The impact of groundwater contaminant migration from a landfill at Site F will depend on the chemical concentrations which occur in the groundwater and the degree to which those concentrations may impair the current or future use of the groundwater, and whether the discharge of contaminated groundwater to surface water will impair its current or future use. Irrespective of any uncertainties in the data, past studies at the Burlington and Bayview Landfill have shown that concentrations of volatile organic chemicals substantially above drinking water standards (ie. US EPA or NYS DEC) do not occur downgradient of the existing landfills. The concentrations of volatile organic chemicals from a landfill at Site F would be expected to be similarly low. In addition, the groundwater in the shale is generally non-potable due to naturally high concentrations of salts and there are no downgradient groundwater users which do not have alternate piped municipal water services. For these reasons, landfilling operations at Site F are not expected to have a significant impact on groundwater.

The impact of the discharge of contaminated groundwater to surface water south of the site will depend on the type of contaminants present, their concentrations, the groundwater discharge rate and the degree of dilution and attenuation which will occur in the surface water environment. Several properties inherent to volatile organic chemicals will contribute to minimize their effect in a surface water environment. In general these compounds are not acutely toxic to aquatic life at concentrations of tens of μg/L and do not bioaccumulate in plants and aquatic organisms. Volatile organics will also be readily lost from the surface water into the atmosphere. These factors will apply to any site. With regard to Site F specifically, the principal factor which will minimize the effect of the discharge of contaminated groundwater

will be the degree of dilution which will occur in the surface water and near-surface groundwater environments. The groundwater discharge upward from the shale bedrock toward the ground surface south of the site will occur over a large area, and in comparison to the estimated rate of groundwater discharge (several L per minute or less) over this area the dilution is very large. A key observation in this regard is that if groundwater discharge to the ground surface and surface water were substantial, it should be evidenced by elevated chloride concentrations in the overburden groundwater and surface water because the shale groundwater contains naturally high chloride concentrations. This does not appear to be the case south of Site F, and the difference in chloride concentrations between the shale groundwater (1000's mg/L) and overburden groundwater (100's mg/L) suggest a dilution factor of at least ten to one hundred times. This degree of dilution would also act on volatile organic contaminants emerging in any groundwater discharge from the shale.

Based on the observations of limited groundwater contaminant migration from the existing Burlington and Bayview Landfills, the existing surface water and groundwater conditions down-gradient of the landfills which will provide for substantial dilution of the discharge of any landfill-derived contaminants, and the contaminant migration simulations which suggest long travel times despite not accounting for matrix diffusion, it is my opinion that the migration of groundwater contaminants from the proposed Site F will not have a significant impact on groundwater or surface water quality down-gradient from the site.

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STANLEY FEENSTRA

EDUCATION:

1978	B.Sc. (Honours Co-Operative), Applied Earth Sciences, University of Waterloo
1980	M.Sc., Hydrogeology, University of Waterloo
1987	Enrolled in part-time Ph. D. program in Hydrogeology at the University of Waterloo. Supervisor: Dr. John A. Cherry

PROFESSIONAL EXPERIENCE:

1987- Hydrogeochemist and President, Applied Groundwater Research Ltd., Mississauga, Ontario.

Responsible for advice to CIBA-GEIGY and its consultants regarding the investigation and remediation of dense non-aqueous phase liquid (DNAPL) chemicals at a Superfund site in Pennsylvania.

Responsible for advice to an electronics manufacturer and its consultants regarding groundwater contamination by organic solvents at plant sites in New York.

Responsible for advice and expert testimony for the US EPA and Dept. of Justice regarding groundwater contamination by TCE at a manufacturing facility in Pennsylvania.

1985-1987 Senior Hydrogeochemist, ZENON Environmental Inc., Burlington, Ontario

Responsible for sampling and analysis program for groundwaters and soils for the New York Power Authority at a former chemical plant site in Niagara Falls, N.Y.

Responsible for sampling, equipment proofing and analysis of deep groundwaters from Paleozoic rock formations in Niagara Falls, Ontario for Environment Canada.

Responsible for the evaluation of enhanced dissolution methods for the remediation of dense non-aqueous phase liquid (DNAPL) chemicals from the subsurface for Environment Canada.

Responsible for the evaluation of the chemical aspects of groundwater contaminant migration at the candidate sites for the proposed Region of Halton landfill.

1980-84 Contaminant Hydrogeologist, Golder Associates, Mississauga, Ontario

Responsible for a comprehensive review of the co-disposal of hazardous wastes with municipal wastes into landfills for Environment Canada and Environment Ontario.

Involved in investigations of potential soil and groundwater contamination at a variety of petroleum refining, distribution and marketing facilities, and at commercial hazardous waste disposal facilities in Alabama, Colorado and New York.

Responsible for the preparation of a comprehensive manual for the sampling of uranium mine tailings for publication by the National Uranium Tailings Program of Energy, Mines and Resources Canada.

Responsible for the design and directed the installation and monitoring of state-of-the-art instrumentation to assess the in situ performance of the 1.2 metre thick engineered clay liner at the Keele Valley Landfill, Maple, Ontario.

Involved in the review and assessment of various chemical disposal facilities in Niagara Falls, New York for Canadian and U.S. regulatory agencies.

1978-80 Research Associate in Hydrogeochemistry, Department of Earth and Sciences, University of Waterloo.

Responsible for an Atomic Energy of Canada Limited research project on the evaluation of the geochemical evolution of sulfate in groundwater using stable isotopes.

1975-77 Environmental Technician, Ontario Ministry of the Environment.

Involved in the collection and evaluation of surface water and sediment quality data from the Great Lakes and inland waters of eastern Ontario.

PROFESSIONAL ACTIVITIES:

Association of Ground Water Scientists and Engineers
American Chemical Society
American Geophysical Union
International Association of Hydrogeologists

PUBLICATIONS AND PRESENTATIONS:

1. Feenstra, S., P. Fritz and E.J. Reardon (1979) "The Chemical and Isotopic Evolution of Sulfur in a Shallow Groundwater Flow System, Chalk River, Ontario". Presented at the Perch Lake Study Symposium/Workshop, October 3-4, 1979, Chalk River Nuclear Laboratories, Atomic Energy of Canada Limited, Chalk River, Ontario.

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