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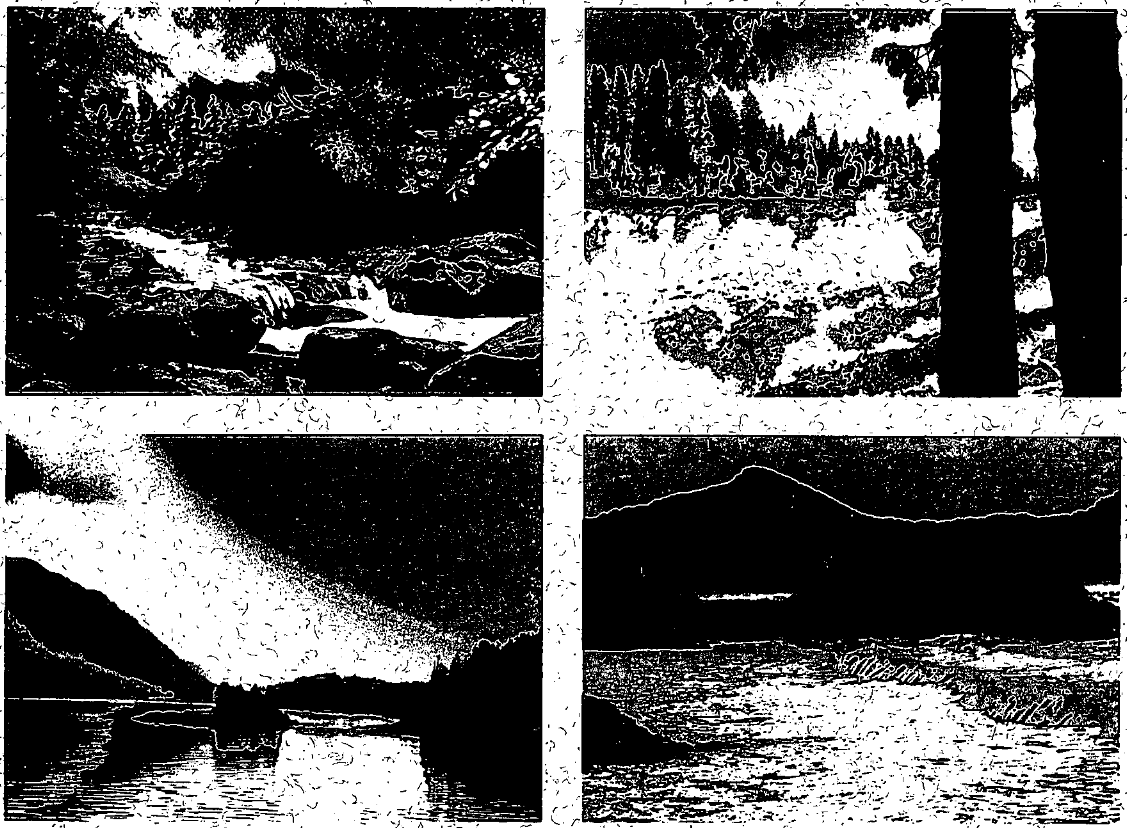
2. Remedial Investigation

R.E. Jackson, S. Lesage, M.W. Priddle, A.S. Crowe and S. Shikaze



SCIENTIFIC SERIES NO. 181

INLAND WATERS DIRECTORATE
NATIONAL WATER RESEARCH INSTITUTE
CANADA CENTRE FOR INLAND WATERS
BURLINGTON, ONTARIO, 1991

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Abstract

Hydrogeochemical research by NWRI scientists during the period 1986-89 has shown that a large plume of organic chemicals has continued to migrate through an outwash aquifer following disposal at the Special Waste Compound (SWC) at the Gloucester Landfill, near Ottawa, Ontario. The leading edge of the plume, characterized by high concentrations of chloride and aliphatic ethers, e.g., 1,4-dioxane, has moved off site. Most of the halogenated hydrocarbons have not been transported so far and remain directly below the SWC in the outwash aquifer and the overlying aquitard. In this zone there is considerable evidence of dehalogenation reactions, which are probably microbially catalyzed.

Field and laboratory experiments were undertaken to determine the mobility of the aliphatic ethers and some halogenated hydrocarbons in the outwash aquifer. The individual retardation factors derived from these experiments show good correlation with the octanol-water coefficient of each compound. Comparison of the laboratory and field experiments indicates that laboratory soil columns yield reasonable estimates of the retardation factors determined in the field.

The remedial alternatives considered for the site are reviewed and preliminary remedial measures taken on behalf of the site owner, Transport Canada, are described. Containment systems, such as slurry walls, are unsuitable because of the permeable nature of the bedrock beneath the SWC. The principal remedial option recommended by consultants to Transport Canada should be successful in decontaminating large parts of the aquifer; however, the aquitard underlying the SWC may be extensively contaminated by nonaqueous phase liquids and therefore require excavation. In situ biorestoration may be required to complete aquifer decontamination following the five-year pump and treat operation.

Résumé

Des recherches hydrogéochimiques effectuées entre 1986 et 1989 par les scientifiques de l'INRE ont montré qu'un grand panache de produits chimiques organiques poursuit sa migration dans un aquifère alluvionnaire par suite de l'élimination à l'enceinte réservée aux déchets spéciaux (EDS) du site d'enfouissement de Gloucester près d'Ottawa (Ontario). Le bord d'attaque du panache, caractérisé par des concentrations élevées de chlorures et d'éthers aliphatiques (p. ex. 1,4-dioxane), a franchi les limites du site. La plupart des hydrocarbures halogénés n'ont pas été transportés si loin et restent directement sous l'EDS dans l'aquifère alluvionnaire et la couche semi-perméable sus-jacente. Il y a dans cette zone des indications considérables sur l'existence de réactions de déshalogénation probablement catalysées par l'action microbienne.

Des expériences ont été entreprises sur le terrain et en laboratoire afin de déterminer la mobilité des éthers aliphatiques et de certains hydrocarbures halogénés dans l'aquifère alluvionnaire. Les facteurs particuliers de retard dérivés de ces expériences présentent de bonnes corrélations avec les coefficients de partage octanol-eau de chacun des composés. La comparaison des expériences effectuées en laboratoire à celles menées sur le terrain indique que les colonnes de sols en laboratoire fournissent une évaluation raisonnable des facteurs de retard sur le terrain.

Les solutions de dépollution envisagées pour le site sont examinées et les mesures préliminaires de dépollution appliquées pour le compte du propriétaire du site, Transports Canada, sont décrites. Les systèmes de confinement, comme les murs de ciment, ne conviennent pas en raison du caractère perméable du socle rocheux sous l'EDS. La principale option de dépollution recommandée à Transports Canada par les spécialistes devrait permettre de réussir à décontaminer de grandes parties de l'aquifère; toutefois, la couche semi-perméable sous l'EDS peut avoir subi une contamination étendue par les liquides de la phase non aqueuse et son excavation pourrait s'avérer nécessaire. Une biorestauration sur le site pourra être nécessaire pour compléter la décontamination après l'opération de pompage et de traitement d'une durée de cinq ans.

Abbreviations

CFC113:	1,1,2-trichloro-1,2,2-trifluoroethane
DCA:	dichloroethane
DCE:	dichloroethene
DEE:	diethyl ether
DNAPL:	dense, nonaqueous phase liquid
EPA:	U.S. Environmental Protection Agency
GAO:	U.S. General Accounting Office
GEC:	A.J. Graham Engineering Consultants
GC/MS:	gas chromatography/mass spectrometry
K_{ow}:	octanol-water partition coefficient
LNAPL:	light, nonaqeous phase liquid
M:	multilevel, as in 54M
MSD:	mass selective detector
P:	piezometer, as in 134P
PAH:	polyaromatic hydrocarbon
PCE:	perchloroethylene or tetrachloroethene
QA:	quality assurance
QC:	quality control
R_f:	retardation factor
RI/FS:	remedial investigation/feasibility study
SIM:	selected ion monitoring
SWC:	the special waste compound at the Gloucester Landfill
TCA:	1,1,1-trichloroethane
TCDD:	tetrachlorodibenzodioxin
TCE:	trichloroethene
TCM:	trichloromethane or chloroform
THF:	tetrahydrofuran
VC:	vinyl chloride or chloroethene

CHAPTER 1

Introduction

Public concern over the health risks associated with hazardous waste sites and the consumption of ground waters possibly polluted by these sites has become acute in the past decade. The assessment and remediation of such sites has involved the expenditure of millions of dollars of public and private funds in Canada and many times that amount in the U.S.A. With the announcement of the recent federal/provincial agreement (April 1989) to create a five-year, C$250 million hazardous waste site cleanup program for orphan sites throughout Canada, it is essential that reliable and complete hydrogeologic remedial investigations must be undertaken if the program is to avoid many of the inefficiencies that have plagued the US $5 billion Superfund of the U.S. EPA (e.g., Freeze and Cherry 1989; Hirschhorn and Oldenburg 1988). This report is an example of the kind and detail of hydrogeologic and hydrogeochemical information that will be required at Canadian hazardous waste sites if effective and efficient remedial technologies are to be chosen.

The purpose of this report is first to review and update the hydrogeological and hydrogeochemical data from the Gloucester Landfill, near Ottawa, Ontario, and second to assess the remedial options for the restoration of the outwash aquifer beneath the Special Waste Compound (SWC). The extent of ground-water contamination at this site has previously been documented in Jackson *et al.* (1985), i.e., Part 1 of this report. In reviewing the hydrogeochemical data and remedial options, reference will be made to the reports of the various consultants, notably Water and Earth Sciences Ltd. (WESA) and Intera Technologies Ltd., who have undertaken hydrogeological studies either for the owner of the property, Transport Canada, or for Environment Canada. However, this report is primarily a presentation and interpretation of data from 1988 collected by the National Water Research Institute (NWRI) group associated with the continued assessment of ground-water contamination arising from disposals of organic liquid wastes within the Special Waste Compound at the Gloucester Landfill.

The remainder of the first chapter is concerned with the origin and evolution of ground-water contamination at hazardous waste sites, the identity of the contaminants of concern at other sites in eastern Canada, the health risks associated with the consumption of such contaminants in ground waters, and the need for a reliable, remedial investigation of the hydrogeology of a site.

Chapter 2 contains a review of waste disposal, hydrogeological, and hydrogeochemical data from the Gloucester Landfill from previous reports that are not now generally available. New hydrogeochemical data collected by NWRI during 1988 are presented in Chapter 3 together with a discussion of the sampling and analytical methods used. These new data when coupled with the information on contaminant mobility presented in Chapter 4 allow the hydrogeologist to assess the appropriateness of the various remedial options (Chap. 5).

GROUND-WATER QUALITY AT HAZARDOUS WASTE SITES IN EASTERN CANADA

It is becoming increasingly apparent that ground-water quality at most hazardous waste sites is dominated by the occurrence of light and/or dense (relative to water) nonaqueous phase liquids, i.e., LNAPLs or DNAPLs. While it is entirely possible that sites contain neither NAPL, i.e., the source of all contaminants is an aqueous solution, the potential presence of NAPLs requires that this possibility be investigated before any remedial option is chosen.

Figure 1 shows the location of five hazardous waste sites in eastern Canada that have been investigated by the NWRI ground-water contamination group. DNAPLs have been identified in ground-water samples and by fouled downhole equipment at three of the five hazardous waste sites listed in Table 1; their presence in the other two sites is suspected but not confirmed. In addition to DNAPLs, samples from

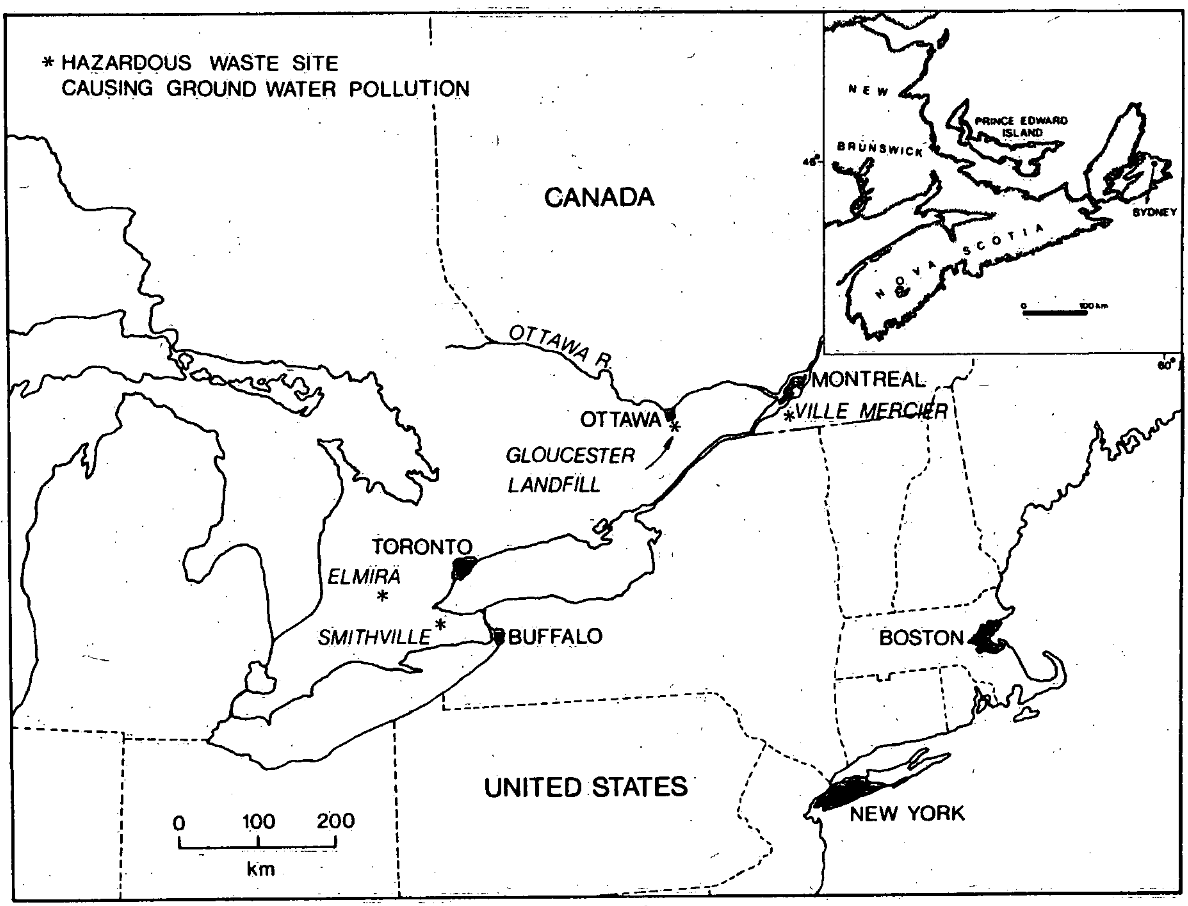

Figure 1. Location of five hazardous waste sites in eastern Canada.

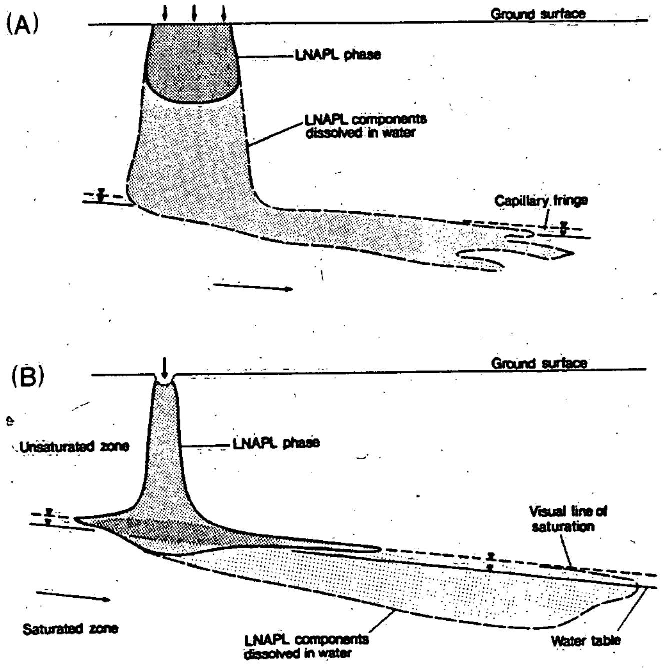

Figure 2. LNAPL migration patterns (after Schwille 1975).

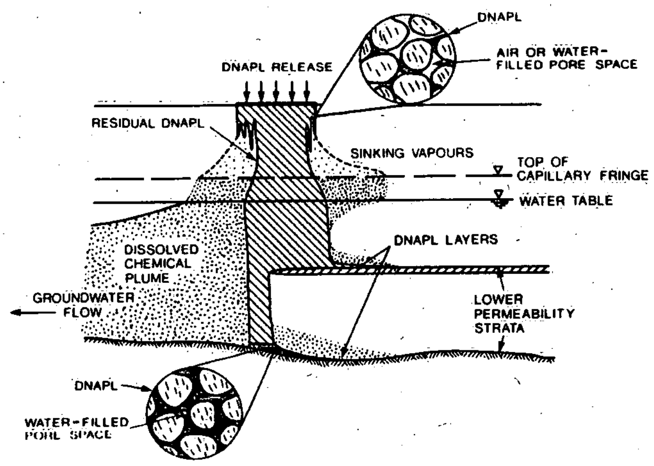

Figure 3. DNAPL migration patterns (after Feenstra and Cherry 1988).

monitoring wells at Sydney, Nova Scotia, contained LNAPL liquids.

Ground waters migrating around NAPL pools or through parts of aquifers containing residual NAPL emulsions, i.e., droplets or "ganglia," will become contaminated by mass transfer of the NAPLS to the ground water. As Hunt *et al.* (1988a) have shown, such mass transfer is very slow and dependent upon the solubility and molecular diffusivity of the organic compounds comprising the NAPL, the rate of ground-water flow past the NAPL zones, and the surface area of the NAPL. The processes by which sinking LNAPL and DNAPL plumes dissolve and contaminate a whole aquifer is shown in Figures 2 and 3.

The concentration of the "i"th component of the NAPL in ground water is given by the approximation (Stumm and Morgan 1981; Banerjee 1984)

$$C_i = x_i C_s$$

where x_i is the mole fraction of the "i"th component in the NAPL and C_s is the aqueous solubility of the component. Two important observations may be made about this expression.

First, the equilibrium concentration of the "i"th contaminant in ground water adjacent to a NAPL pool will be less than the aqueous solubility limit by a factor proportional to the mole fraction of the contaminant in the pool. This is because the contaminant preferentially partitions into the pool (Reinhard *et al.* 1984). Second, in the case of several contaminants being present in a NAPL pool with similar mole fractions, the concentration of each in adjacent ground water will be a function of their respective aqueous solubility. With time, the more soluble components will preferentially dissolve in the ground water because such mass transfer is primarily a function of aqueous solubility rather than ground-water flow rate (Anderson *et al.* 1987). This phenomenon is observed in the weathering of oil seeps (Hunt 1979) and in the remediation of solvent plumes in ground water (Feenstra, S., 1988, Applied Ground Water Research Ltd., Mississauga, Ont., pers. comm.).

Volatile halogenated aliphatics, pesticides, and aromatic hydrocarbons have dissolved in ground water from NAPL pools and/or ganglia at most of the sites listed in Table 1 and have created plumes with aqueous concentrations of tens of mg/L, as shown in Table 2. Because these concentrations exceed drinking water limits by as much as one thousand fold, the presence of these contaminants is of foremost concern.

Table 1. Five Hazardous Waste Sites in Eastern Canada

Site	Geology	Source of contamination
Elmira	Sequence of sand and gravel aquifers separated by till aquitard.	Industrial lagoons with chemical process wastes.
Gloucester	Sequence of sand and gravel aquifers separated by till aquitard.	Disposal of 100 000 L of lab solvents, wood preservatives, and inorganics.
Mercier	Esker overlying fractured bedrock.	Disposal of 40 000 000 L of liquid industrial wastes.
Smithville	Thin till over fractured dolomite.	Industrial reprocessing of organic liquids.
Sydney	Thin till over fractured sandstone.	Coking coal manufacture and coal tar production.

Table 2. Concentration Ranges of Contaminants of Concern in Ground Waters at Five Hazardous Waste Sites in Eastern Canada (ppm (>1 mg/L), ppb (≈ 1-1000 μg/L), and ppq (= pg/L) indicate maximum levels detected in several monitoring wells at each site.)

Site	DNAPL composition	Contaminants of concern
Elmira	Presence uncertain	BTEX, phenols and CPs (ppm); 2,4-D and 2,4,5-T (ppb), PCDD (soil)
Gloucester	Presence uncertain	CFC113, solvents, VC, dioxane, benzene (ppb); PCDDs, and PCDFs (ppq)
Mercier	Oil + PAHs + solvents	Solvents, benzene, CB, and VC (ppm)
Smithville	Oil + TCE + PCB + CBs	PCBs, TCE, and CBs (ppb)
Sydney	Coal tar	BTEX and naphthalene (ppm)

Abbreviations:

BTEX = benzene, toluene, ethyl benzene, and xylene
CBs = chlorobenzenes
CPs = chlorophenols
dioxane = 1,4-dioxane
CFC113 = chlorofluorocarbon-113 (1,1,2-trichloro,1,2,2-trifluoroethane)
PCB = polychlorinated biphenyls
PCDD = polychlorinated dibenzodioxins
PCDF = polychlorinated dibenzofurans
solvents = tetrachloroethene (PCE), trichloroethene (TCE), 1,1,2-trichloroethane (TCA), 1,2-dichloroethane
VC = chloroethene (vinyl chloride)
2,4-D = 2,4-dichlorophenoxyacetic acid
2,4,5-T = 2,4,5-trichlorophenoxyacetic acid

HEALTH RISKS OF CONTAMINATED GROUND WATER

Guideline values for Canadian drinking water quality are set by a federal/provincial subcommittee. Some of the more relevant values are shown in Table 3 along with comparable U.S. EPA values. In most cases, Canadian values do not exist for those compounds identified in the plumes migrating from the hazardous waste sites listed in Table 1; consequently U.S. EPA values are frequently adopted by Canadian regulatory authorities.

The setting of these values is done in an extremely conservative way, involving the extrapolation of animal cancer tests and sometimes human epidemiological data to predict human health risks. Information from the most detailed assessment of these risks yet published (Gold *et al.* 1984) is displayed in schematic form in Figure 4, which includes a carcinogenic rating of a wide variety of compounds, most of which have been identified in the five hazardous waste sites considered in this paper. While this procedure has been criticized by leading toxicologists as being too conservative (Ames *et al.* 1987; Smith 1988), it is nevertheless a methodology that is widely used by public health agencies. Thus, the health risk estimation technique forces chemists and hydrogeologists to design sampling and analysis programs to identify and quantitate potential carcinogens in ground water at concentrations that comply with the guideline values, the so-called compliance levels.

The risk associated with drinking ground water contaminated with any of the compounds shown in Figure 4 is assumed proportional to the value of

Table 3. Drinking Water Quality Guideline Values in µg/L (ppb)

Compound	Canadian recommended maximum acceptable concentration	U.S. EPA recommended maximum contaminant level
Benzene	5	5
Chloroethene (VC)	n.d.	2
1,4-Dichlorobenzene	5	75
1,2-Dichloroethane	n.d.	5
1,1-Dichloroethene	n.d.	7
Tetrachloromethane	5	5
1,1,1-Trichloroethane (TCA)	n.d.	200
Trichloroethene (TCE)	n.d.	5

Sources: Guidelines for Canadian Drinking Water Quality, 1987, and Federal Register, 52(130): 25690-25717.

n.d. = Not defined.

TD_{50} – "the dose rate (in mg/kg body weight/day) that, if administered chronically for the standard lifetime of the species (e.g., male rats), will halve the probability of remaining tumorless throughout that period" (Gold *et al.* 1984). This index of carcinogenic potency ranges from the most toxic – an isomer of tetrachlorodibenzodioxin (2,3,7,8-TCDD) – to the least – green food dye no. 1. However, the contaminants of greatest concern to the hydrogeologist are those that are not only toxic but also mobile and persistent in the subsurface (Jackson et al. 1985).

Consequently, TCDD isomers, which are occasionally found in soils and ground water, may be of less concern at a particular hazardous waste site because of their very low mobility due to their low solubility. Most of the organic contaminants of concern identified in Table 3 occupy that region in Figure 4 between 1 mg/(kg· d) and 1 g/(kg· d). The compound with the lowest drinking water guideline value in Table 2 is vinyl chloride or chloroethene, which is a known human carcinogen and which is produced in ground water by the biodegradation of several common chlorinated organic solvents (Wolf *et al.* 1987). Because of its combined toxicity, mobility, and persistence, vinyl chloride is the contaminant of greatest concern at many sites, even though these sites may contain compounds identified in Figure 4 as being of much greater carcinogenic potential. This fact necessitates the undertaking of remedial investigations that are optimized for such contaminants of concern.

THE NEED FOR RELIABLE REMEDIAL INVESTIGATIONS

The assessment of hydrogeological and hydrogeochemical conditions at a hazardous waste site should lead to a realistic appraisal of the potential of the various remedial technologies to achieve the required corrective action. In American parlance these two steps are known as the remedial investigation and feasibility study or RI/FS when they are issued as a single report. Because of the inadequacy of many remedial investigations in the U.S.A. (GAO 1988) and Canada, the prescribed remedial technologies have failed to attain their stated objectives, which may have been unattainable in the first place.

An excellent example of a remedial investigation that failed to identify the actual conditions at a hazardous waste site and thereby undermined the objectives of the feasibility study is that of Ville

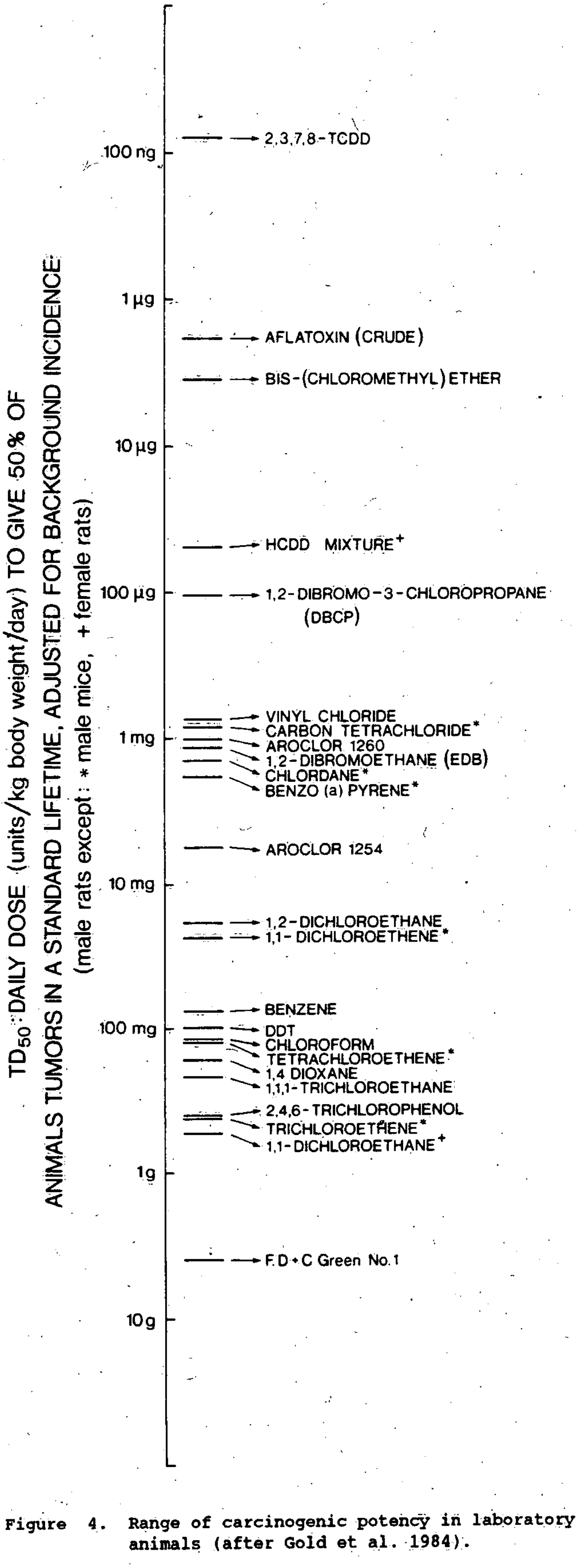

Figure 4. Range of carcinogenic potency in laboratory animals (after Gold et al. 1984).

Mercier, Quebec (Martel 1988). From 1968 to 1972, approximately 40 000 m^3 of liquid industrial wastes were disposed of in lagoons in an abandoned gravel pit near Ville Mercier on the south shore of the St. Lawrence River, 20 km from Montreal (see Fig. 1). By 1971, local wells were closed due to contamination and by 1972, disposals were ordered ended. However, by May 1988, a plume of volatile organic chemicals had migrated at least 4.8 km from the disposal areas and had caused the closure of industrial and municipal water supply wells in Sainte-Martine (see Fig. 5) as well as numerous domestic wells between Sainte-Martine and Mercier.

In 1982, an RI/FS was undertaken that led to the commissioning of a treatment plant in June 1984 to decontaminate the ground water withdrawn from the aquifer beneath the former lagoons. It was intended that the treated water would have "similar characteristics as drinking water" and could be used by Mercier for its potable water supply after five years of pump and treat operations (Simard and Lanctot 1987). This treatment plant and three purge wells operated from June 1984 to June 1985 at 1.2 million m^3/a at a cost of $7.4 million but failed to produce any significant improvement in the ground-water quality entering the treatment plant vis-à-vis the drinking water guidelines. As of August 1988, concentrations of four toxic organic chemicals in the purge wells adjacent to the lagoons were still substantially in excess of drinking water guidelines (Table 4).

The failure of the pump and treat operations to attain the objective of producing drinking water for Ville Mercier after five years of operation was due to the presence of DNAPL pools within the sand and gravel surficial aquifer and the fractures of the bedrock. Chemical analysis of one of these pools by Laval University indicated that it was composed of a complex mixture of refined petroleum hydrocarbons (54%), volatile organic solvents (18%), and polyaromatic hydrocarbons (PAHs) (4.7%) (H. Pakdel 1988, Chemical Engineering Department, Laval University, pers. comm.). Ground water in contact with this DNAPL contained volatile organic solvents in concentrations of only 1% or 2% of their aqueous solubility, indicating relatively slow mass transfer between the pool and nearby ground water. Consequently, aquifer purging operations failed to remove this subsurface source of contamination from beneath the lagoons, with the exception of a pool that was pumped into the treatment plant and fouled the air stripping columns, clarifier unit, and granular activated carbon columns.

Table 4. Ground-Water Chemistry of Selected Toxic Organic Chemicals in µg/L at Two of the Purge Wells, Mercier, Quebec, in August 1988, Compared to U.S. EPA Maximum Contaminant Levels (MCLs) in Drinking Water

Compound	Well 1	Well 3	EPA MCL
Chloroethene (VC)	160	92	2
Benzene	174	162	5
1,2-Dichloroethane	435	723	5
Trichloroethene (TCE)	105	102	5

Sampled and analyzed by NWRI.
MCLs taken from Federal Register, 52(130), p. 25691, 8 July 1987.

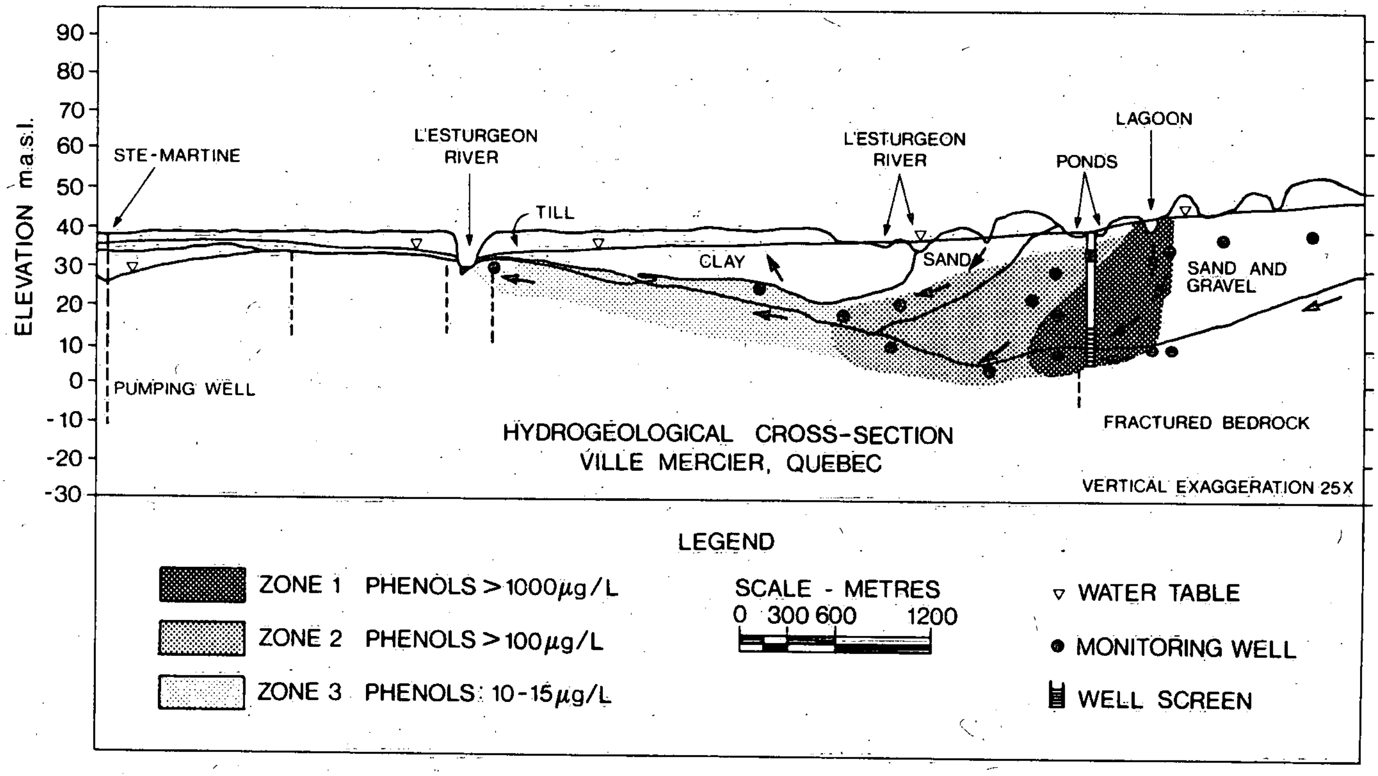

Figure 5. Hydrogeological cross section through surficial and bedrock aquifers, Ville Mercier, Quebec (from original by Poulin et al. 1985).

Because the DNAPLs were not identified in the 1982 RI/FS, the remedial technology, pump and treat, chosen to decontaminate the aquifer could only contain the migration of the dissolved contaminants from the DNAPL but could not remove the DNAPL pools from the aquifer (Martel 1988). This conclusion has also been reached in the U.S.A. in similar situations (Feenstra and Cherry 1988; Mackay and Cherry 1989). Apart from excavation of shallow DNAPL pools, there is no proven remedial technology to remove DNAPLs, although some techniques appear to show great promise (see Chap. 5).

The RI/FS of the Mercier site was completed for $300,000 (Simard and Lanctot 1987), of which less than $100,000 was for hydrogeological and hydrogeochemical investigation. At the Gloucester Landfill, where only 100 m^3 of liquid wastes were disposed of, Transport Canada has spent approximately $450,000 on hydrogeological and hydrogeochemical studies, and the research institutes of Environment Canada (National Hydrology Research Institute, NWRI, River Road Environmental Technology Centre, Wastewater Technology Centre) have spent at least as much. In no sense then can the 1982 Mercier RI/FS be considered to have been reliable and complete, because of the very small budget allocated to the hydrogeological investigation given the scale of disposals at the site.

Three items were particularly absent from the original Mercier RI/FS. Firstly, the sand and gravel aquifer beneath the lagoons was not sufficiently well investigated and therefore the DNAPL pools were not identified. Had they been identified, it would have been possible to determine, by laboratory column experiments, that their removal would be impossible by pump and treat operations. In fairness, it must be added that the DNAPL problem was not fully appreciated by the hydrogeological community in 1982; however, the use of laboratory experiments in contaminant hydrogeology was well established. Secondly, no detailed assessment was conducted of the extent of contamination of the bedrock beneath the sand and gravel. Because of the very low bedrock porosities and high hydraulic gradients, ground-water flow and contaminant transport is extremely rapid in the bedrock fractures and has led to the development of the long plume extending towards Sainte-Martine. Thirdly, no mathematical simulation modelling of ground-water flow and contaminant transport was undertaken to optimize the remedial operations. Mathematical modelling of the site (D. Isabel, 1988, Geological Engineering Department, Laval University, pers. comm.) has now shown that containment of the plume can be accomplished at much lower pumping rates than have been used over the period 1984-89, and therefore containment can be accomplished much more inexpensively due to lower granular activated carbon costs.

The Mercier remediation has been a costly learning experience that should be fully appreciated by Canadian regulators, engineers, and hydrogeologists. While it is clear that not every site can be subject to the same extensive investigation that has characterized the Gloucester site, there must be some balance struck between funds allocated for a particular RI/FS, the health risks posed by the site, and the value of the ground-water resource that is threatened. However, it stands to reason that millions should not be poured into site remediation before a reliable and complete remedial hydrogeologic and hydrogeochemical investigation is performed.

The Gloucester Landfill and Outwash Aquifer

Much hydrogeological and hydrogeochemical work has been undertaken at the Gloucester site during the past decade by both consultants to Transport Canada and research scientists and engineers of Environment Canada (e.g., Jackson *et al.* 1985). This chapter summarizes the hydrogeological and hydrogeochemical results arising from the site assessment studies of NHRI and these consultants from 1976 to 1985.

SITE DESCRIPTION

Between 1969 and 1980, the Government of Canada disposed of hazardous wastes, principally organic chemicals from its laboratories in Ottawa, at the nearby Gloucester Landfill (Fig. 6). The disposal operations took place within a "Special Waste Compound," hereafter referred to as the SWC, located at the southwest corner of the landfill. These operations involved the excavation of trenches (12 m long x 3 m x 3 m) into which bottles of liquid chemical wastes were placed and subsequently combusted by detonation of explosive charges set within them. The wastes consisted of nonchlorinated solvents (56.4 m^3), unspecified wood preservatives (30 m^3), chlorinated solvents (8.6 m^3), and smaller amounts of acidic and ferric chloride and other wastes (GEC 1985a).

HYDROGEOLOGY

Detailed soil coring and hydrogeological testing have resulted in the identification of the five hydrostratigraphic units shown in Figure 7. Unit A, the fractured limestone bedrock, is encountered at depths between 25 and 30 m and is partly covered by a till (Unit B) of relatively low hydraulic conductivity. Unit C, overlying the till and/or bedrock, is a thick (up to 25 m) sequence of silts, sands, gravel, and boulders known as glacial outwash and forms a semiconfined aquifer beneath the landfill. This outwash is overlain by a discontinuous layer of silt, unit D. The surficial aquifer, Unit E, may be up to 10 m thick and is composed of sands and gravels.

The outwash aquifer, Unit C, is composed mineralogically of feldspar (50+%), quartz (20%), and minor amounts of mica, calcite, dolomite, and hornblende. The organic carbon content of the aquifer sediments averages 0.06%. The aquifer has a mean hydraulic conductivity of 10^{-4} m/s, a porosity of 0.30-0.35, and an average linear ground-water flow velocity of 0.05 m/d (Jackson *et al.* 1985; Michel *et al.* 1984). The ground-water temperature is in the range 9-10°C.

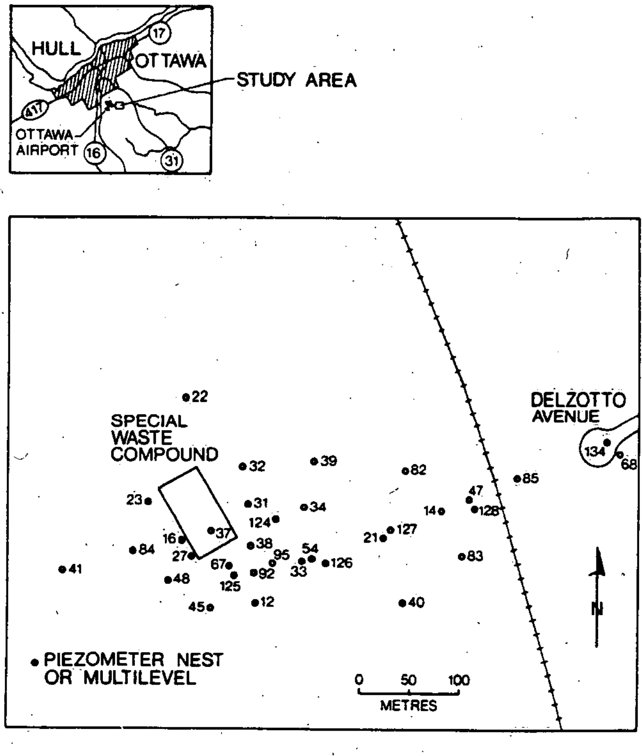

Figure 6. Location of study area near Ottawa and locations of piezometers and multilevels in the monitoring well network. The property boundary of the site is identified by the railway tracks running northwestward across the figure.

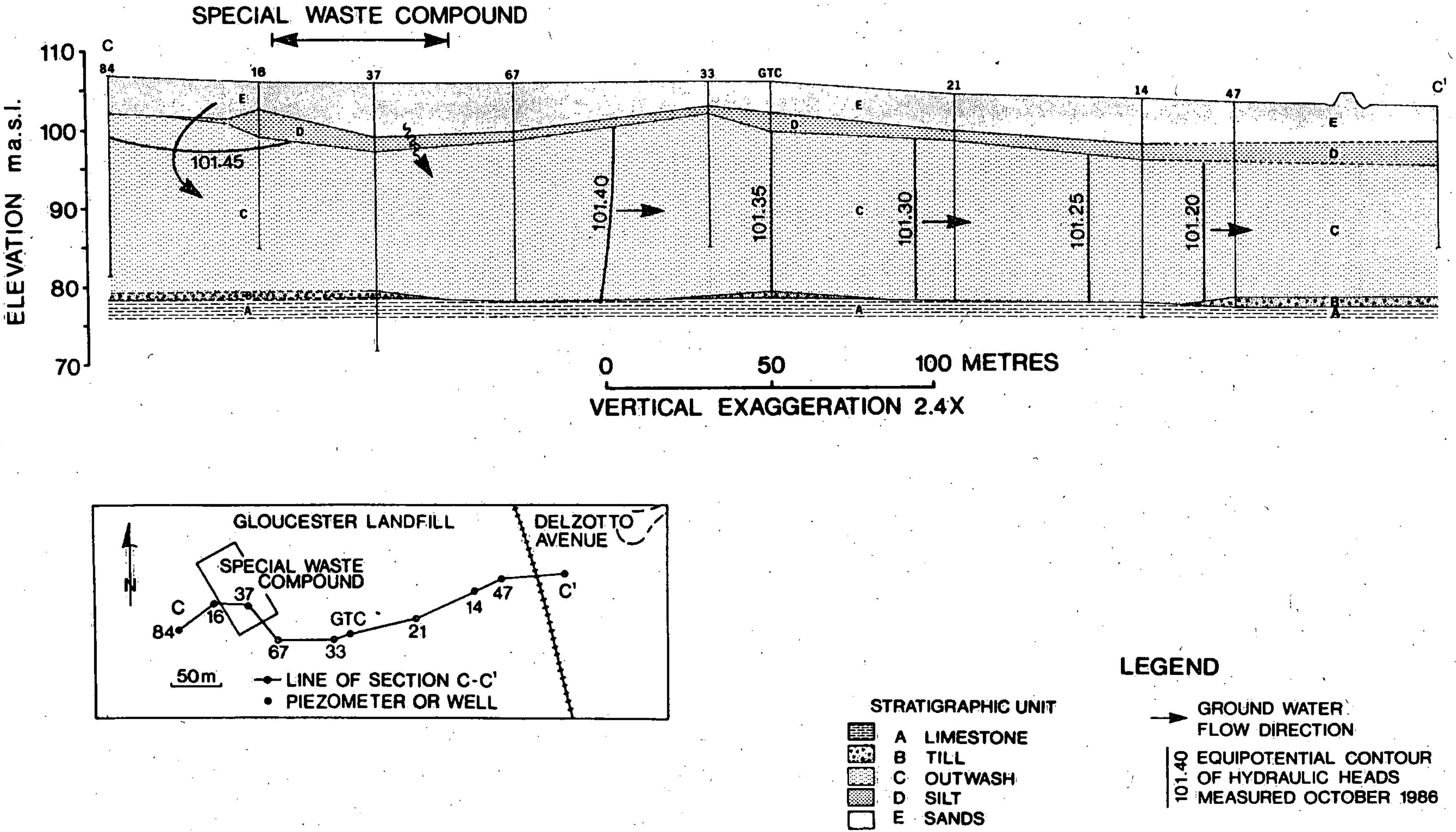

Figure 7. Hydraulic head contours in metres above mean sea level (m.a.s.l.) along the cross section identified in the inset as measured during October 1986. Squiggly arrow beneath the SWC indicates the potential for leakage through confining unit D. Note vertical exaggeration.

The ground-water flow pattern beneath the SWC in October 1986 is shown in Figure 7. Downward ground-water flow occurs from the surficial aquifer, Unit E, to the confined outwash aquifer. Some seepage appears to occur through permeable windows in the silty confining layer, Unit D. Once solutes enter the outwash aquifer, the vertical equipotential lines dictate that they are transported horizontally to the east. Field tests have demonstrated that the underlying bedrock is hydraulically connected to the outwash aquifer (Jackson et al. 1985). Figure 8 shows the flow pattern in the outwash aquifer at the same time but in plan view.

INORGANIC GROUND-WATER QUALITY

The distribution of chloride in plan and cross section in the outwash aquifer is shown in Figures 9 and 10, respectively. The first samples collected at the Gloucester Landfill in November 1977 showed values less than 5 mg Cl⁻/L in that area (see Fig. 9) where concentrations are now in excess of 250 mg Cl⁻/L. Background chloride concentrations in ground waters from upgradient areas west of the SWC (e.g., 41P, see Fig. 6) are less than 10 mg/L. Much higher levels occur beneath and downgradient from the SWC. The vertical configuration of the chloride plume (Fig. 10) along a section extending downgradient from the SWC indicates that chloride is transported downward from the trenches within the SWC into the outwash aquifer and then downgradient to the east. The leading edge of this plume has crossed beneath the railroad tracks near Delzotto Avenue (Fig. 9). This

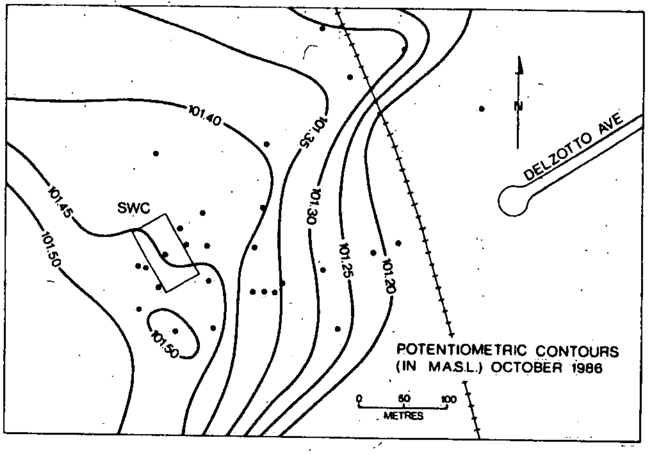

Figure 8. Hydraulic head contours in metres above mean sea level within the outwash aquifer as measured during October 1986.

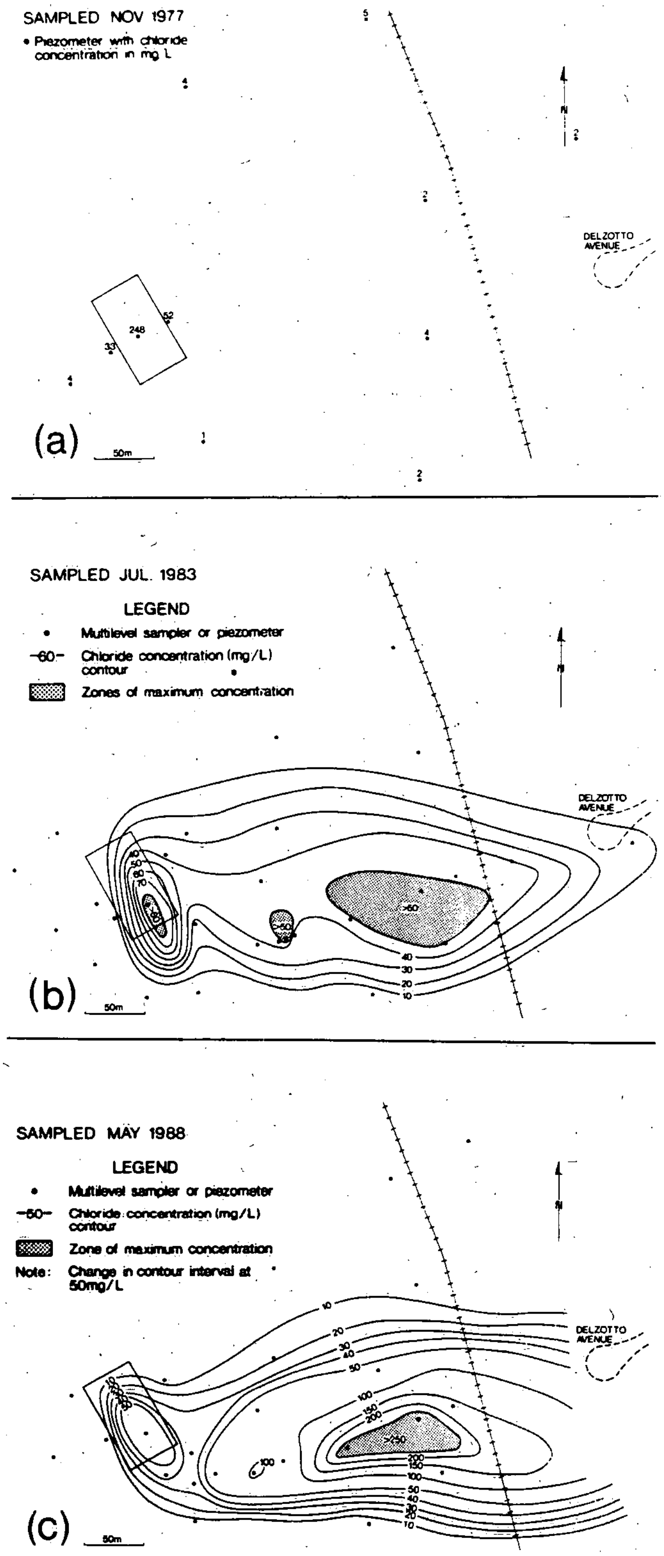

Figure 9. Concentration contours (in mg/L) of the maximum values of chloride from multilevel samplers within the outwash aquifer: (a) November 1977, (b) July 1983, and (c) May 1988. The SWC is outlined at left.

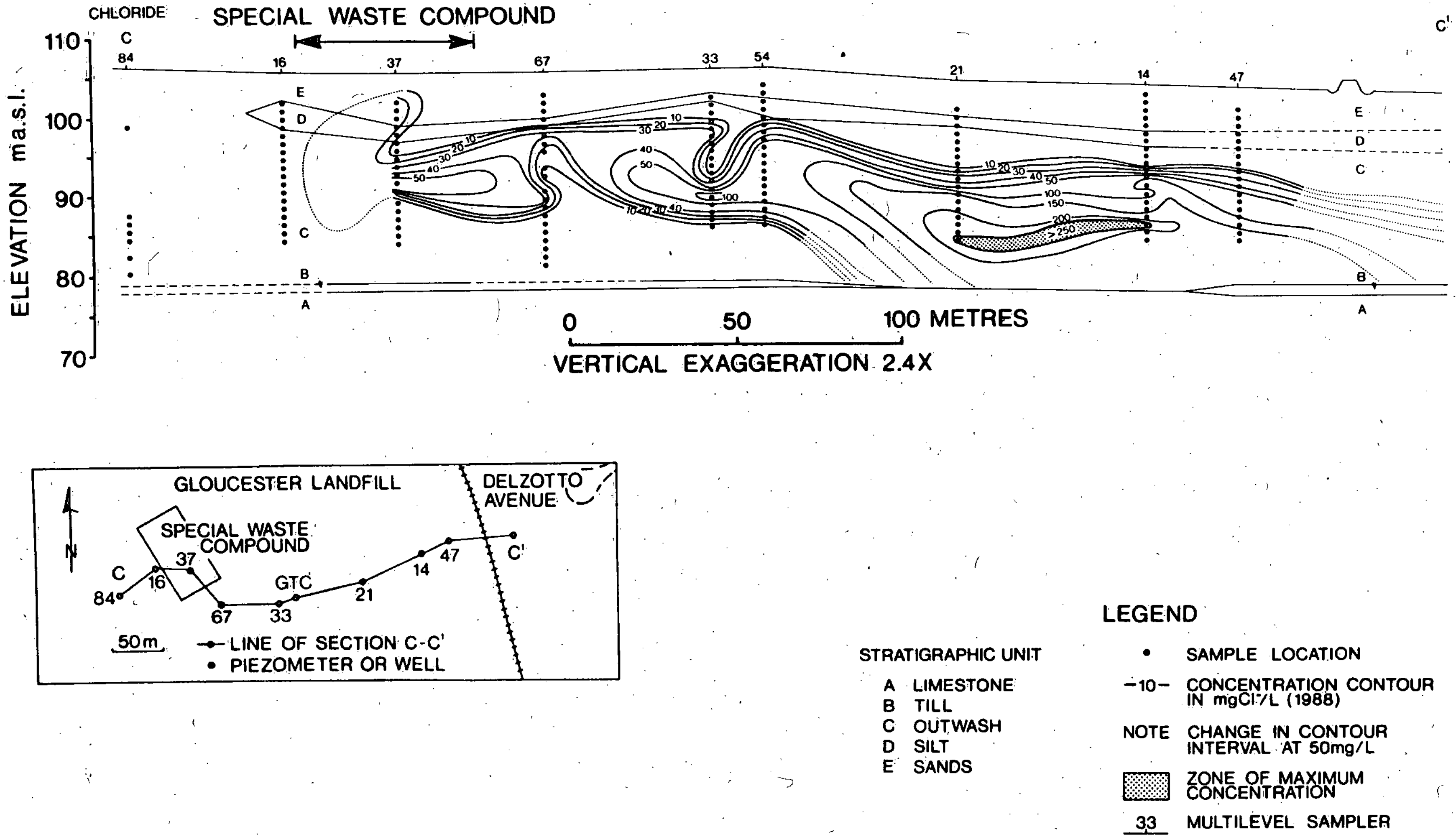

Figure 10. Concentration contours (in mg/L) of the chloride plume along the indicated cross section (see inset), May 1988.

pattern of migration is consistent with the ground-water flow directions shown in Figures 7 and 8. Based on ground-water velocity measurements in the outwash aquifer and assuming no retardation, the plume is advancing at a rate of approximately 5 cm/d.

The pH of shallow ground waters upgradient of the waste site is on the order of 8.0, as shown in Table 5. Because rainfall and snowmelt are generally acidic (pH < 5), considerable neutralization occurs during infiltration and percolation as a result of the dissolution of carbonate minerals, which are a significant component of the aquifer materials. Saturation index data (Table 5) indicate that all ground waters are near or above saturation with respect to calcite and ferrihydrite ($Fe(OH)_3$). Within the chloride plume the pH is generally less than 8.0 and occasionally less than 7.0. This is probably due to the hydrolysis of CO_2 evolved during the oxidation of the organic residues from the SWC, which produces titration alkalinities in excess of 10 meq/L compared with 3-4 meq/L in uncontaminated ground waters (Jackson *et al.* 1985).

Because of its importance in the transformation of organic compounds, the redox state of the ground-water flow system was determined in detail, and has been monitored over an eight-year period. Uncontaminated ground waters upgradient of the SWC (e.g., 41P) contain measurable quantities of dissolved oxygen (2-6 mg/L) and exhibit E_H values greater than 0.5 V. Within the chloride plume, ground waters have nondetectable quantities of dissolved oxygen (i.e., <0.3 mg/L) and E_H values ranging from -0.04 to 0.2 V; such values have been previously associated with sulfate reducing environments in ground-water flow systems (Champ *et al.* 1979; Jackson and Patterson 1982). Ground-water sulfide data for the outwash aquifer measured in 1983 from two multilevel monitors (37M and 67M) are shown in Table 6. Because the highest levels of organic contamination of the outwash aquifer occurred in the period 1982-84, prior to some pumping at well 36W, it is important to note that no significant change in the pH or redox environment has occurred during the period 1982-88 (see Table 7).

Table 5. Inorganic Ground-Water Quality Data at Selected Multilevel Samplers (16M, 37M) and Piezometers (41P) within the Outwash Aquifer (See Figs. 2 and 4 for location of monitors.)

Parameter	Units	41P	16M-16	37M-9
Sp. cond.	µs/cm	380	730	1380
pH		7.9	7.3	7.2
E_H	volts	0.55	0.13	0.13
O_2	mg/L	2.1	≤0.2	≤0.2
Tit. alk.	meq/L	3.6	7.0	10.4
TIC	mg/L	47	99	141
TOC	mg/L	1.3	14	61
Ca^{2+}	mg/L	60	120	210
Mg^{2+}	mg/L	21	20	58
Na^+	mg/L	1.2	13	88
K^+	mg/L	0.9	6.7	3.5
Fe^{2+}	mg/L	0.5	1.5	6.7
Mn^{2+}	mg/L	0.1	n.m.	0.2
SO_4^{2-}	mg/L	32	65	290
Cl^-	mg/L	6.2	27	200
SI calcite		0.34	0.26	0.45
SI siderite		-6.7	0.16	0.76
SI ferrihydrite		3.2	1.1	1.4

Sp. cond. = Specific electrical conductance at 10°C.
E_H = Measured Pt electrode potential referred to normal hydrogen electrode.
Tit. alk. = Titration alkalinity.
TIC = Total inorganic carbon.
TOC = Total organic carbon.
n.m. = Not measured.
SI = Saturation index, log IAP − log K_{sp}.

Table 6. E_H and Dissolved Sulfide in Ground Waters from the Outwash Aquifer Directly beneath the SWC

Monitoring point*	E_H (V)	Sulfide (mg/L)
37M-1	0.07	0.01
37M-6	0.06	0.12
37M-7	0.06	0.13
37M-9	0.04	0.075
67M-6	n.m.	0.005
67M-7	n.m.	0.005
67M-9	0.05	1.1
67M-11	n.m.	0.040

*See Fig. 3 for location of monitoring points.
n.m. = Not measured.

Table 7. pH, E_H, and Dissolved Oxygen in Ground Waters (monitor 37M-9) beneath the SWC in 1982, 1983, and 1988

	1982	1983	1988
pH	7.2	7.0	7.1
E_H(V)	0.13	0.04	0.11
O_2 (mg/L)	<0.3	<0.3	<0.3

ORGANIC CONTAMINANT MIGRATION

In addition to chloride contamination of the outwash aquifer, Jackson *et al.* (1985) reported substantial organic contamination. The patterns of organic contaminant migration in the outwash aquifer in 1982 resulting from disposals in the SWC suggest that the organic solvents have undergone chromatographic dispersion, i.e., aqueous-phase solute transport concurrent with sorption of the contaminants, principally by organic carbon particles within the aquifer sediments (Patterson *et al.* 1985). As shown in Figure 11, the organic contaminants have been transported distances that are inversely proportional to their respective octanol-water partition coefficients. Because the octanol-water partition coefficient (K_{ow}) provides a measure of a compound's propensity to become sorbed by organic matter (i.e., hydrophobic partitioning), it follows that the observed transport of the organic solvents by ground water is in accord with the hydrophobic sorption theory of Schwarzenbach and Westall (1981) and others. This pattern of behaviour has implications for aquifer decontamination. Those contaminants that have been transported furthest (i.e., the most mobile ones) will be most easily removed from the aquifer by purging. The remedial system, therefore, must be designed to recover a variety of contaminants with widely varying mobilities.

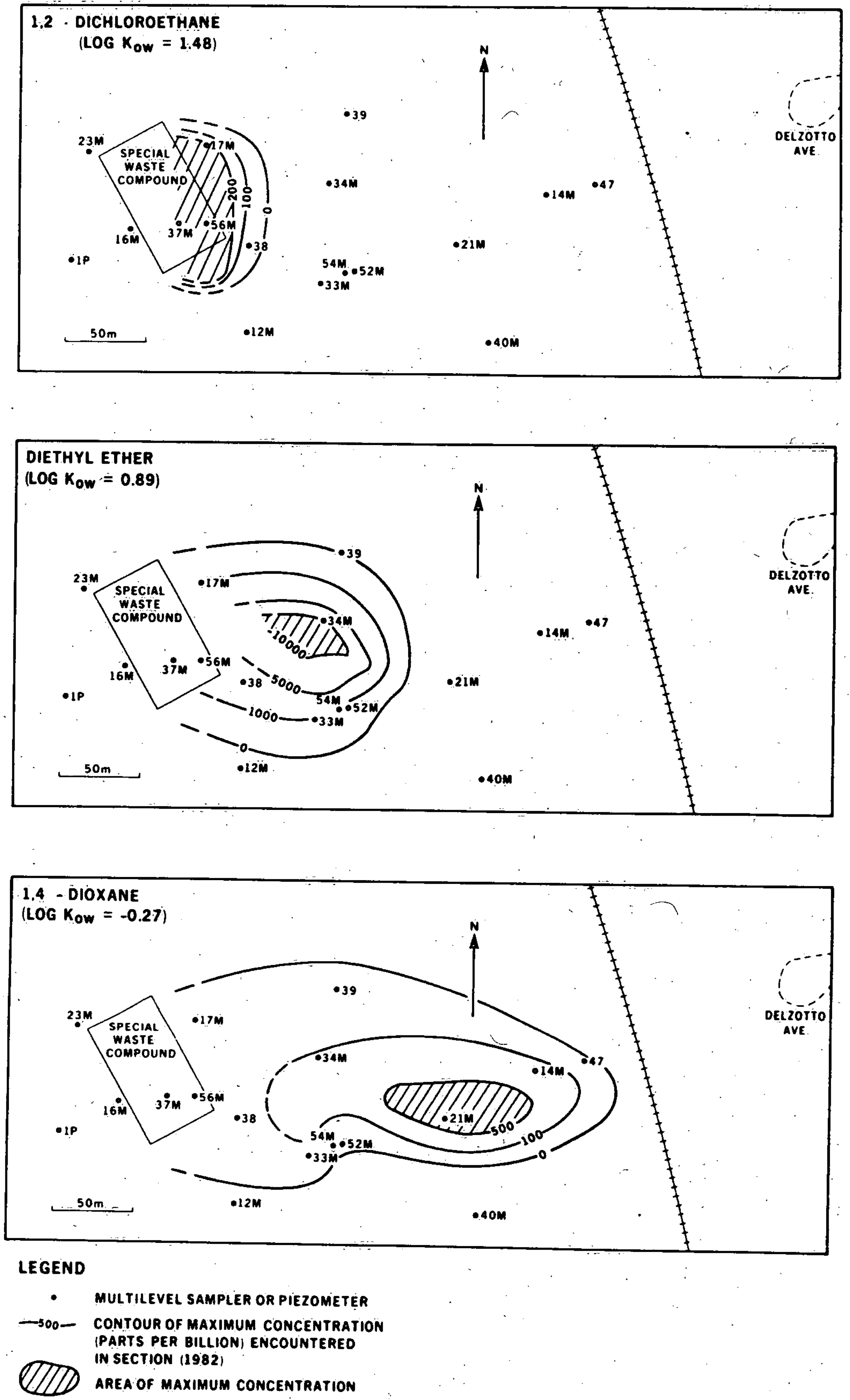

Figure 11. Plan view of three plumes of volatile organic chemicals in 1982 illustrating chromatographic dispersion (from Patterson et al. 1985).

Analysis and Fate of Organic Priority Pollutants in the Outwash Aquifer

The 1982 survey of the Gloucester site by NHRI indicated that there was extensive chloride and organic solvent contaminant migration towards the property boundaries as shown in Figures 9, 10, and 11. Although some heavy metal salts were disposed of in the SWC, detailed hydrogeochemical sampling indicated that they were not present in the ground water in significant amounts (Jackson *et al.* 1985). Consequently, the contaminants of concern at the site have been the EPA list of organic priority pollutants, a group of 129 toxic chemicals identified by Keith and Telliard (1979), some of which are shown in Figure 4 and known also to be carcinogenic. Therefore, detailed annual surveys of organic pollutants were conducted in 1984, 1987, 1988, and 1989. The results for 1984 and 1988 are tabulated in the Appendix.

This chapter presents the general principles followed by NWRI in the sampling and analysis of organic priority pollutants and a discussion of the hydrogeological and hydrogeochemical results of the 1988 survey conducted by NWRI.

SAMPLING AND ANALYSIS OF ORGANIC CONTAMINANTS IN GROUND WATER

Sampling Methods

The sampling of organically polluted ground waters requires a careful choice of monitoring instruments, the principal criterion being that individual hydrostratigraphic units within a ground-water flow system must be individually sampled. Wells that penetrate more than a single unit provide little useful information (Grisak *et al.* 1978; Reilly *et al.* 1989). This is partly because the sample dilution that occurs in fully penetrating wells implies dispersion greater than actually takes place and also because this integrated sample may well indicate contaminant concentrations within the acceptable guidelines, although these guidelines may be exceeded within a particular unit.

Figure 12 shows two commonly used devices for sampling ground-water quality — the bundle-type multilevel sampler and the 5-cm i.d. piezometer. Both provide the capability of sampling small zones of potentially contaminated ground water. The first is frequently used to map the outlines of contaminant plumes in three-dimensional detail (e.g., Jackson *et al.* 1985). The second is generally used for monitoring ground-water quality where it is necessary to establish that this quality is in compliance with regulated or guideline values; consequently the well screen is made of an inert material, e.g., stainless steel.

At the Gloucester Landfill, samples for compliance monitoring have been collected using dedicated, submersible pumps (QED Systems, Ann Arbor, Mich.) with PTFE (Teflon) bladders operated by compressed air or nitrogen that does not come into contact with the ground-water sample. The pumps, shown in Figure 12, are located at the depth of the well screen and can be isolated from the stagnant water in the well bore by inflating a packer system immediately above the pump. Generally, at least three well screen volumes are then pumped before sampling begins (Barcelona and Helfrich 1986; Robin and Gillham 1987).

Figure 13 shows the sequence of operations conducted in the field to collect samples and the subsequent distribution of aliquots for analytical purposes. Samples are collected in precombusted amber glass bottles, with no headspace for volatile organic samples, at a delivery rate of 100 mL/min or less. The bottles are allowed to overflow by at least 1.5 volumes, and then are rapidly capped and stored at about 4°C until analyzed (Barcelona *et al.* 1985). Sample bottles should not be rinsed out with the sample because of the potential for films of NAPLs to coat the bottles.

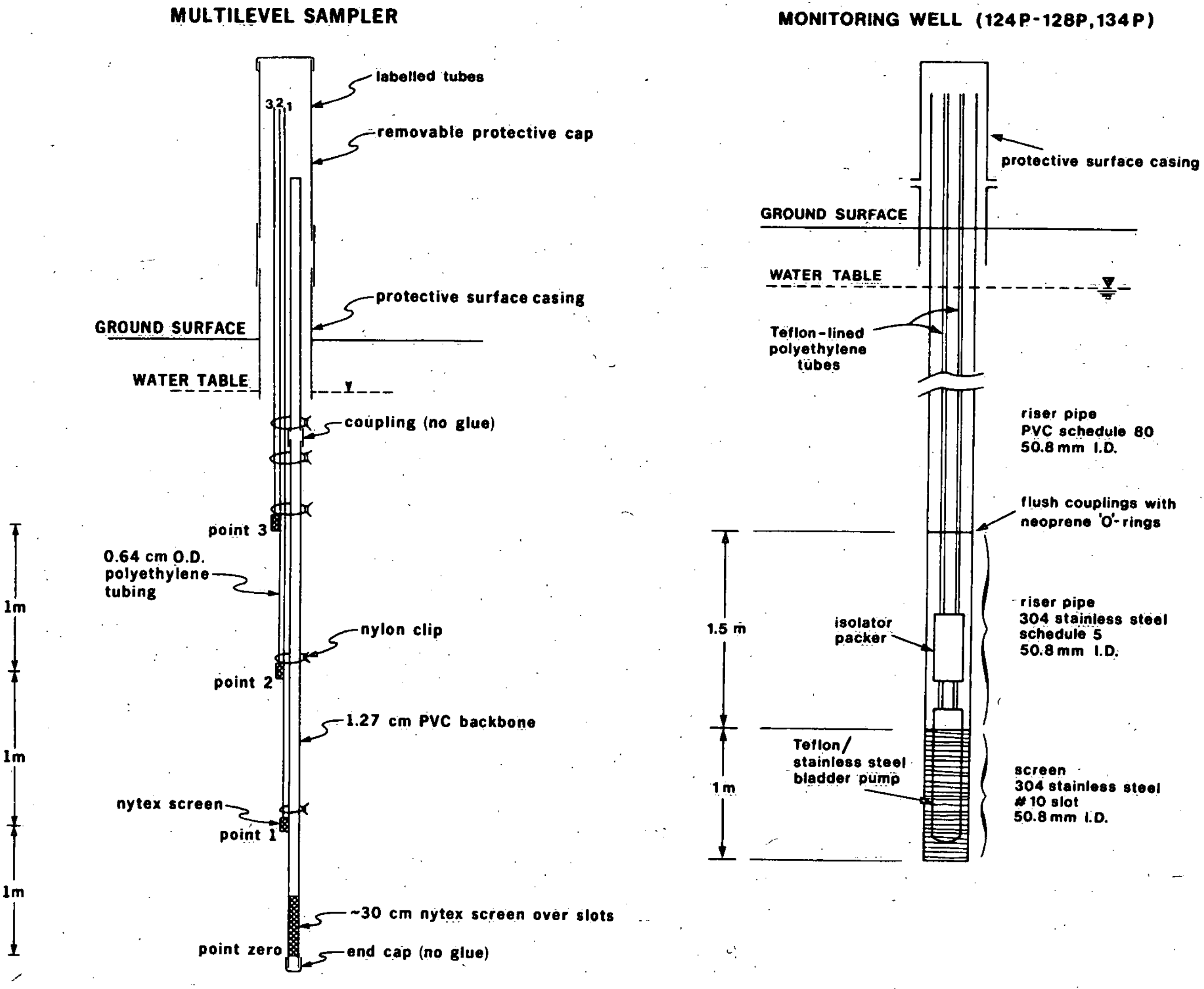

Figure 12. Types of monitoring instruments: (a) the bundle-type multilevel and (b) the 5-cm i.d. piezometer with a dedicated submersible pump.

Preservation techniques for organic constituents of ground water differ from those for inorganics. In particular, it is inadvisable to subject samples collected for volatile organic analysis to vacuum filtration because of the potential volatilization of analytes. However, samples for analysis of volatile aromatic hydrocarbons, e.g., BTEX in Table 2, should be preserved with HCl acid (U.S. EPA 1984a).

Analytical Organic Chemistry

The contaminants of concern at Gloucester were all organic, consequently their analysis at NWRI was by coupled gas chromatography/mass spectrometry (GC/MS). The principal components of GC/MS systems are shown in Figure 14. Essentially, the ground-water sample containing a mixture of chemicals was first concentrated (purge and trap) and then separated into individual chemicals in the capillary column of the GC, each compound yielding a peak on the chromatograph indicating a different retention time within the GC column. The chemicals elute from the GC and are then ionized into fragments of characteristic mass producing mass spectra that allow the identification of each chemical. Note that GC alone cannot confirm the identity of a chemical because many chemicals have similar retention times in GC columns; consequently MS is an essential step in compound identification.

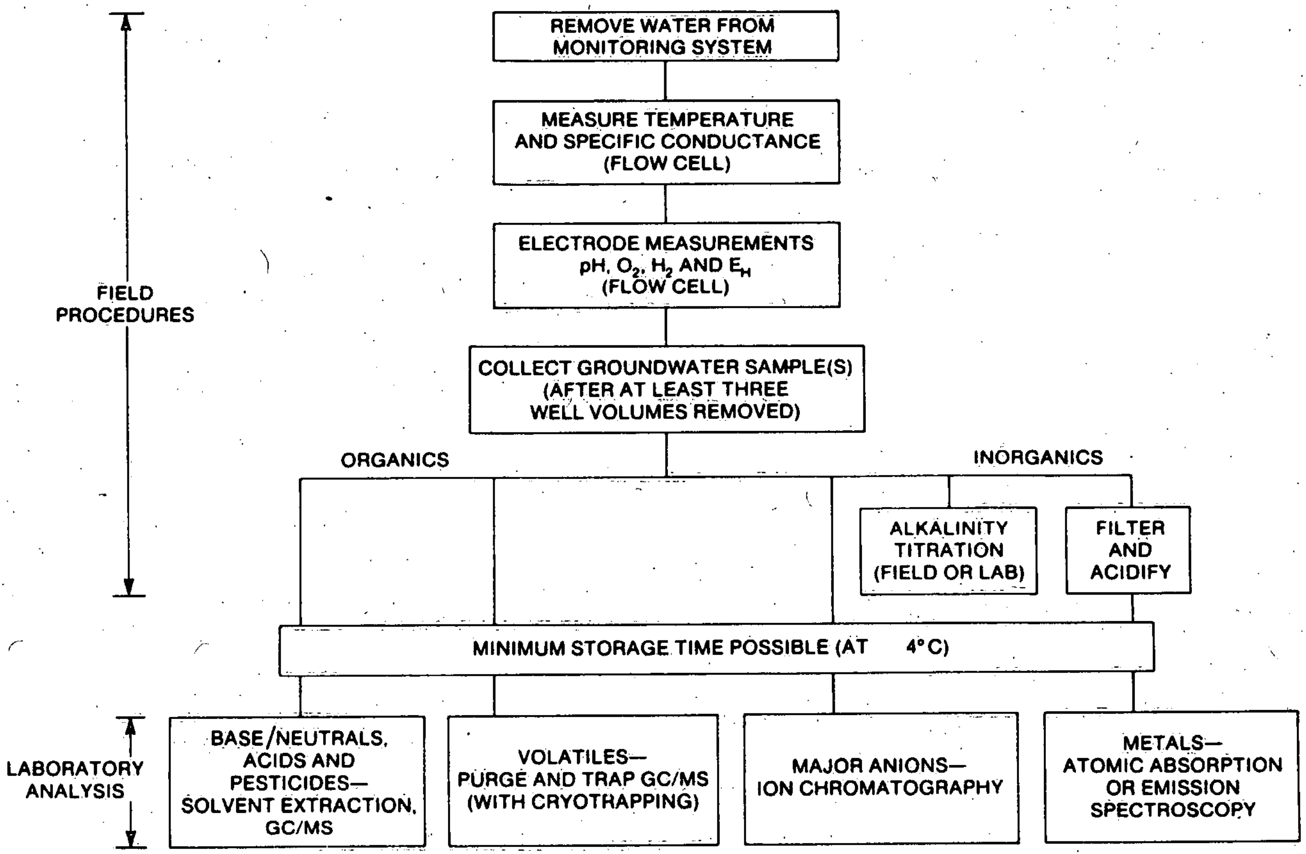

Figure 13. Flow chart of sampling and analytical methods.

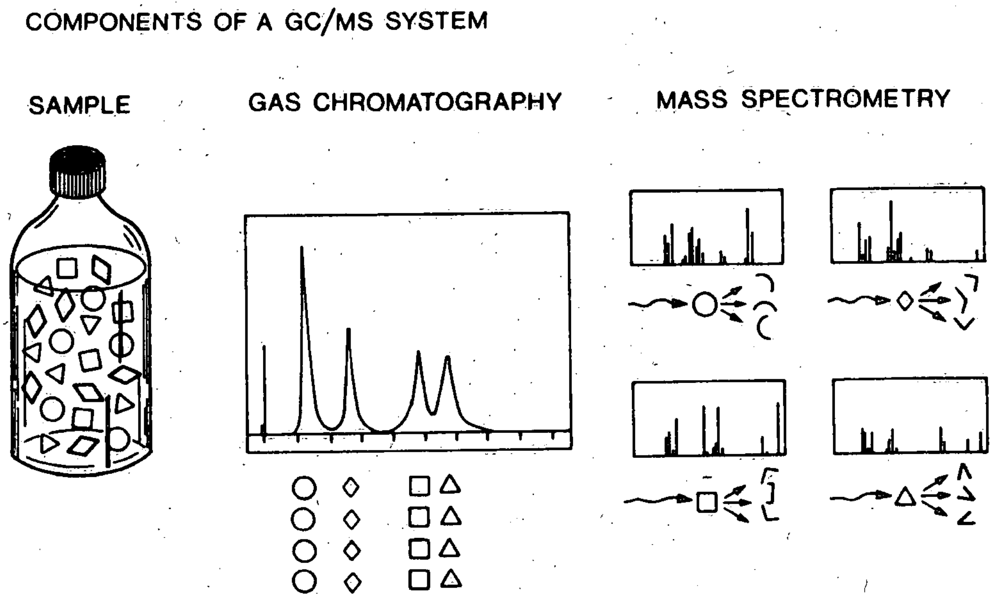

Figure 14. Components of GC/MS system (from Swallow et al. 1988).

GC/MS analysis may be done in two modes — full scan or selected ion monitoring (SIM). In the first case, a range of masses (typically 45 to 450 atomic mass units) is acquired at the rate of one scan per second, i.e., every second a full mass spectrum is obtained and stored in the GC/MS's dedicated computer for later interpretation. In the second case, only those ions characteristic of the contaminants of concern are acquired, therefore SIM maximizes the sensitivity and selectivity of the GC/MS system. The analyses at Gloucester were done in the full scan mode, which allowed the identification of several unsuspected compounds (see below). Figure 15 shows the full scan or total ion current chromatogram for volatile organics in a sample of ground water from the Gloucester Landfill.

Even when a full spectrum is acquired, quantitative analysis is done on extracted ions. One ion is selected for each compound for quantitation, i.e., each ion is a specific detector. The area under the curve of each peak of the chromatogram is integrated and compared to that of a standard. In addition, the area of one or two other characteristic ions is also integrated and their ratios compared to that of the primary ion; these are known as qualifying ions because they allow qualitative identification of the compound. Under the same operating conditions of the GC/MS, the ratios of qualifying to primary ions are constant. These, together with the retention times of the compounds in the GC column, comprise the criteria used to ensure the correct identification of the analytes.

The identification of the compounds contained within a sample is accomplished by comparing the spectrum of the unknown to spectra contained within a library stored within the GC/MS computer. There are two methods of comparison — forward and reverse searches. In a forward search, the spectrum of the

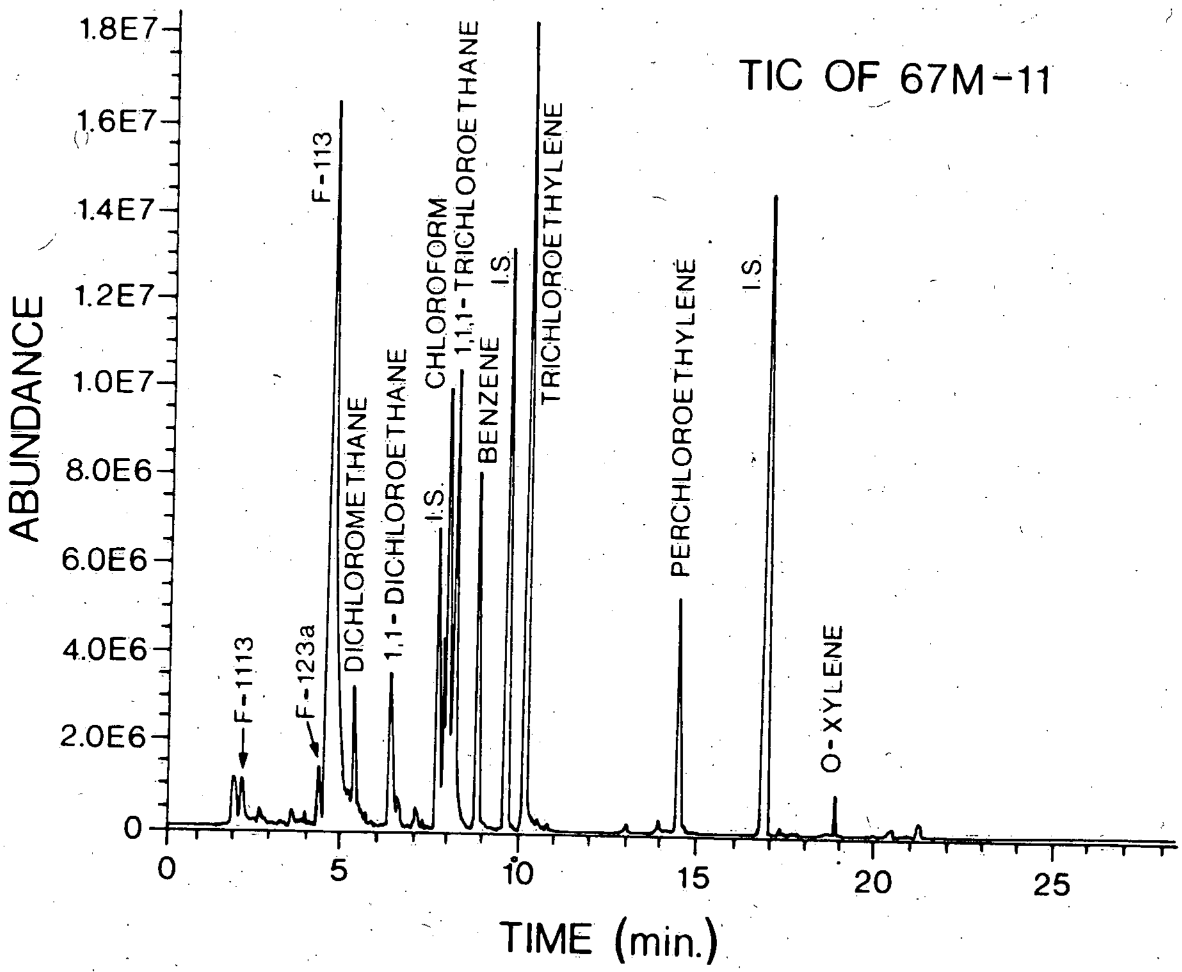

Figure 15. Total ion current chromatograms for multilevel sampling point 67M-11 at the Gloucester Landfill.

unknown is compared to those contained in a library, e.g., that of the U.S. National Bureau of Standards (NBS) or a user-prepared library. The search yields a list of matches ranked in order of best fit. It is not uncommon to have matches of 90-95%, which would be considered confirmatory. However, when the matches are much less, the chemist must interpret the differences and make tentative identifications. In a reverse search, a group of spectra contained in a user-created library is compared to all the spectra found in the sample. This type of search is commonly used in target compound or "priority pollutant" analysis.

Forward searches will also identify nonpriority pollutants that may be of considerable value in identifying the origin or pathway of the contaminated ground water (Swallow *et al.* 1988); however, they are more time consuming and therefore more costly than reverse searches. The latter have the advantage of being able to identify much lower quantities of a compound than is possible in a forward search.

Consequently, the advantages of the mass spectrometer as a detector over other detectors (ECD, FID, etc.) are that the quantitation is not affected by coeluting compounds and that the spectrum allows the unambiguous identification of the target compounds and tentative identification of nontarget compounds.

Quality Assurance/Quality Control

The purpose of any quality assurance (QA) program is to reduce analytical measurement errors to agreed upon limits (quality control or QC) and to ensure that the analytical results have a high probability of being of an acceptable quality, a system known as quality assessment (American Chemical Society 1983).

QA/QC programs need to be operated at two different levels. The first is within the laboratory to ensure that good lab practice is continually employed and that the results achieved are of consistent and acceptable quality. Secondly, external control by the client has to be implemented to ensure comparability of data obtained from different sources and over different time periods.

Two principal factors need to be assessed by the QA/QC program — precision and accuracy. They are defined as follows:

> *Precision:* The degree to which data generated from replicate or repetitive measurements differ from one another (American Chemical Society 1983).
>
> *Accuracy:* The degree of agreement of a measured value with the true or expected value of the quantity of concern (Taylor 1987).

Accuracy is more difficult to assess because, in many cases, certified reference materials, in particular for volatile organic compounds, are not available for many analytes and sample matrices (i.e., the totality of the sample, e.g., landfill leachate) and, furthermore, interlaboratory studies are infrequent. A suitable alternative would be to have another laboratory prepare a set of standards and spiked samples to be included with the rest of the samples. It should be stressed that split samples give little or no information about either precision or accuracy.

The principal QA/QC tools used by NWRI in its assessment of the Gloucester Landfill were the employment of blanks, replicates, surrogates, and internal standards. As a rule, field or trip blanks were collected to evaluate external contamination of the sample, e.g., benzene from gasoline stored near the sample bottles, and the results corrected on the basis of such blanks. Replicates are important because they are the only measure one has of sample variability. Therefore, it is preferable to analyze one sample in triplicate and thus obtain statistics on the variability than to analyze two samples in duplicate, for which statistics cannot be estimated.

Surrogates, compounds that are analogs of the analytes and which are added to the sample prior to the extraction step, have replaced spiked samples as QA/QC tools. This is because the laboratory does not usually know the expected concentration of the analyte a priori, and therefore cannot spike the sample to a meaningful level. Most surrogates are deuterated (deuterium isotope labelled) analogs of the analyte of interest or chemical analogs, e.g., bromodichloromethane for volatile organics, assumed unlikely to be present in the sample. Since the surrogate is added before extraction, it is carried through the entire analytical procedure and its recovery reported as a measure of matrix effect and analytical error.

The final QA/QC tool used at NWRI was that of internal standards. These too may be deuterated analogs; however, they are added in the final step and are used for quantitation. Their purpose is to account for instrument variability from one sample to the next. Thus the peak area of the internal standard in the sample chromatogram is compared to that in the standardization chromatogram and all concentrations are corrected for any variability. For volatile analysis, surrogates and internal standards are added together before the purging step.

RESULTS OF 1988 SAMPLING SURVEY

Sampling

Ground-water samples were collected during May 1988 from selected monitoring points within the monitoring well network. These points are shown in cross section in Figure 16. A set of five samples was collected from 5-cm i.d. piezometers (denoted by 124P...128P), each completed with stainless steel screens and dedicated Teflon bladder pumps and inflatable packers (QED Environmental Systems Inc., Ann Arbor, Mich.), which were inflated above the well screens to contact a stainless steel pipe extension of the screen. Before sampling, at least three well-screen volumes were discharged to waste. The remainder of the samples were obtained from bundle-type multilevel samplers (denoted by 16M, 37M, etc., Cherry *et al.* 1983) using peristaltic pumps. Samples were collected in 25- or 40-mL amber glass vials and kept on ice until reaching the laboratory.

Measurements of pH and E_H were conducted in the field using combination glass and Pt electrodes fitted into airtight flow cells (Jackson *et al.* 1985). Dissolved oxygen was determined using an Orbisphere Laboratories (York, Maine) 2606 oxygen meter. Sulfide measurements were performed using a glass/sulfide electrode couple in acidic solution ($pH < 5$) under closed, continous flow conditions (Gulens 1985).

Analytical Methods

Volatile organic compound analysis was carried out following EPA method 624 (U.S. EPA 1984b). A model 810 Unacon (Envirochem Inc., Kemblesville, Pa.) purge and trap was used to concentrate the analytes. This unit was interfaced to a Hewlett-Packard model 5970 GC-MSD. The analytical column was a 30-m DB-624 (J&W Scientific), 0.32-mm i.d., 1-µm film thickness, directly interfaced into the source of the mass selective detector (MSD). The gas chromatograph was cooled to -5°C with liquid CO_2 to allow the analyses of the purgeable gases and programmed in two ramps to 140°C (-5°C to 35°C at 10°C/min and then at 4°C/min). Difluorobenzene and chlorobenzene-d5 internal standards were added to samples of 10 mL immediately before purging. Mass acquisition was from 40 to 250 a.m.u. at two scans per second, but quantitation was done on extracted ions specific to each analyte, using one qualifier ion for confirmation. Analytical standards were prepared from the purest available chemicals purchased from a variety of sources. Unknowns for which no standards were available were tentatively identified by comparison of their spectra to an NBS library containing 42 000 compounds.

Before 1983, most of the data were obtained using an HP7675 purge and trap interfaced to a GC-FID, with some of the analyses being confirmed on a Finnigan 4000 mass spectrometer. From 1984, a combination of FID and Hall detectors was used, until 1987, when the GC-MSD system described above was acquired.

Semivolatiles and dioxins were analyzed following the EPA 625 and 613 protocols (U.S. EPA 1984c, 1984d), except that a 10-L sample was extracted for dioxins. The analysis was done by Dr. R. Lao, RRETC, Ottawa, on a Finnigan 4000 GC/MS.

Semivolatile Organic Compounds

Because most disposals of liquid organic chemicals at the site involved their incineration, and because pentachlorophenol wastes were thought to be included (i.e., as wood preservatives), the presence of chlorinated dibenzodioxins (CDDs) and dibenzofurans (CDFs) in the ground water was suspected. The most prominent species detected in multilevel 37M-19 were octachlorodibenzodioxin and heptachlorodibenzofuran (Fig. 17). No tetrachlorodibenzodioxins were detected. The total concentrations for CDFs and CDDs were 43 and 59 pg/L respectively. The distribution pattern was similar to that observed for combustion of chlorinated organic products (Czucza and Hites 1984).

Several GC/MS analyses were made in 1983 to detect semivolatile contaminants directly beneath the SWC. Very few compounds were identified, most of which were naturally occurring fatty acids. The only priority pollutant identified was phenol at approximately 90 µg/L in 37P-1. An independent laboratory reported the pesticides silvex and 2,4-D at approximately 20 µg/L in 36W. More recently, analysis of multilevel

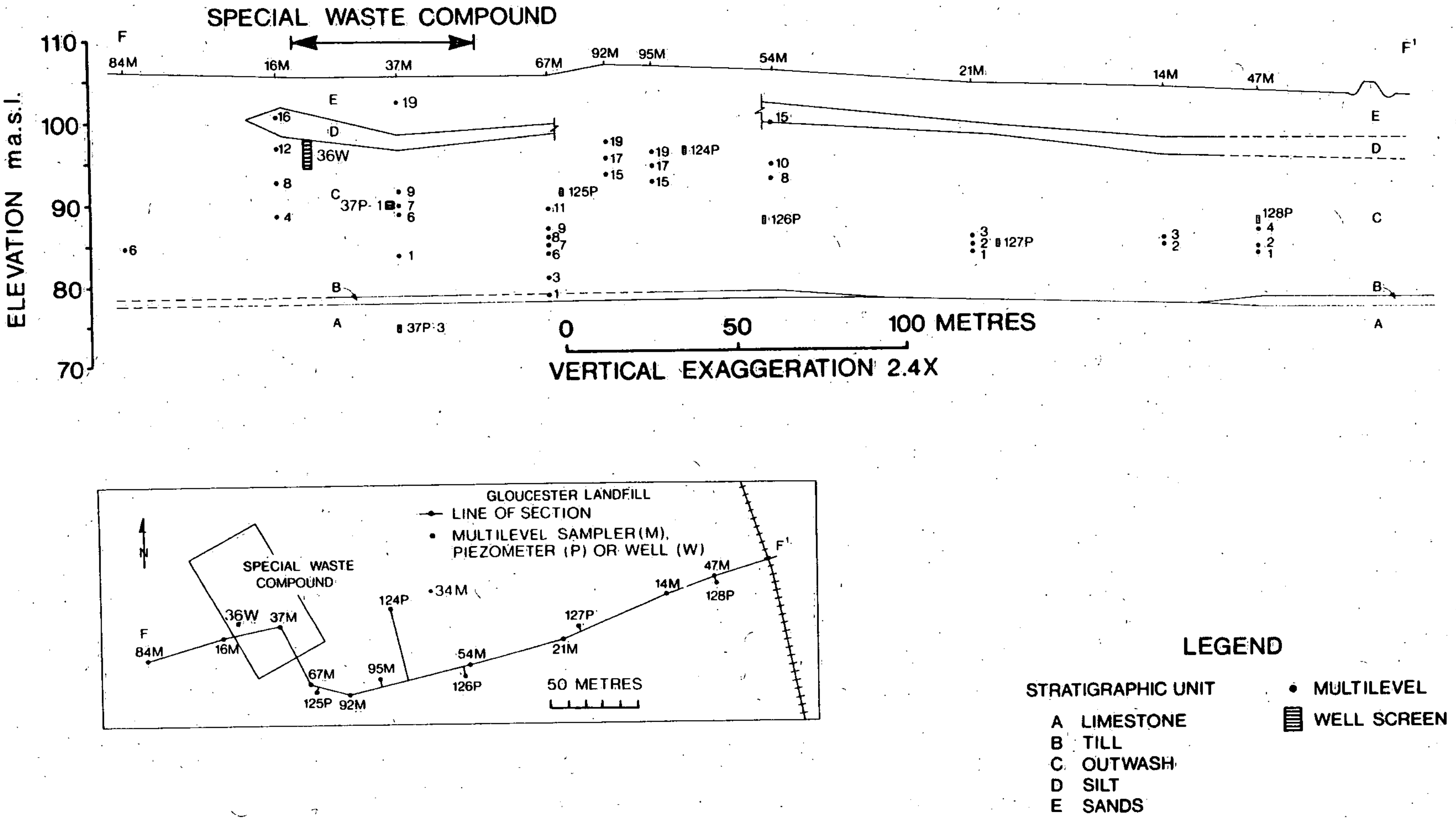

Figure 16. Vertical location of multilevel inlet tubes and monitoring well screens along the cross section identified in the inset. As shown in the inset, some wells off the cross section are projected onto the line of section so as to indicate their depth.

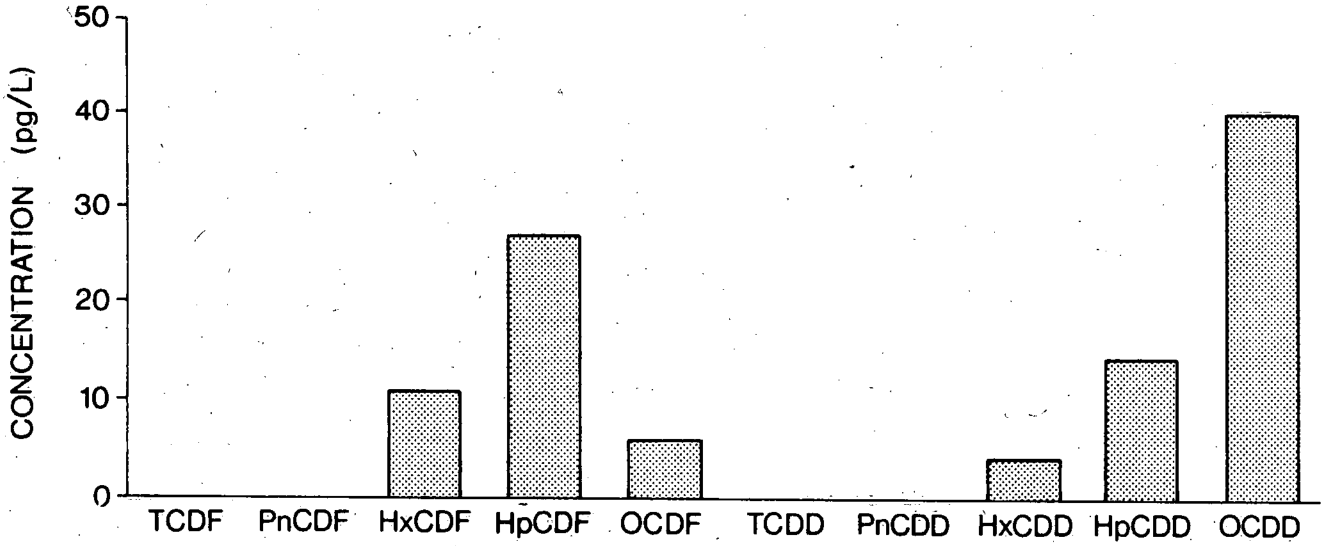

Figure 17. Congener profiles of chlorinated dibenzo-dioxins (CDD) and -furans (CDF) in a sample of ground water from 37M-19. The prefixes are as follows: T = tetra, Pn = penta, Hx = hexa, Hp = hepta.

sample 92M-15 using a dynamic thermal stripper (purge and trap at 95°C) did not reveal any anthropogenic semivolatile compounds. This is not entirely surprising given the nature of the waste and also given the fact that most of these compounds have relatively high log K_{ow} values and would not be expected to migrate very far.

Volatile Organic Compounds

Volatile organic compounds, most of them clearly identifiable laboratory solvents that survived incineration, constitute the bulk of the contaminants at Gloucester and are listed in Tables 8 and 9. Their behaviour will be discussed by groups, according to their mobility, with special attention paid to potential biotransformation products and their distribution. The possibility of the presence of a nonaqueous phase plume will also be discussed. Absolute concentrations (see Table 10) were generally much lower in May 1988 than during previous sampling periods (see Jackson *et al.* 1985) because of the large volume of contaminated ground water pumped from well 36W, used for the assessment of alternative treatment technologies and returned to the aquifer through 36W (Fig. 16). These operations only affected ground-water quality directly beneath the SWC, where they caused a dilution of concentrations. Figures 18 and 19 show the migration patterns of selected volatile organic compounds.

Aliphatic Ethers

These compounds are of particular concern because of their rapid migration in the aquifer (Jackson *et al.* 1985; Patterson *et al.* 1985) as demonstrated by their presence in 126P and 127P (Table 10). The centre of mass of the diethyl ether and tetrahydrofuran plume moved from 34M to 21M from 1982 to 1988. The centre of mass for dioxane moved from 21M in 1982 to 47M in 1988. The movement of dioxane is of special concern because it has been recognized as a carcinogen. Although the U.S. EPA has not officially classified this chemical, the drinking water concentration estitimated to increase cancer risk by 10^{-6} is of 7 µg/L (U.S. EPA 1988). The current analytical methodology (U.S. EPA 1987) only achieves a practical detection limit of 150 µg/L, making the monitoring of the dioxane plume difficult. Also, this compound seems to have been refractory to biodegradation in the anoxic environment.

Analysis of samples collected from 85M, 68M, and 134P during May 1989 indicates that a dioxane plume with maximum concentrations in excess of 1 mg/L has moved off site beneath the railroad tracks.

Aromatic Hydrocarbons

The concentration of these contaminants was very low, except for benzene. Therefore, it is most likely that there were no petroleum products other than laboratory solvents disposed of at the site, since one would then expect to find a larger number of alkyl benzenes. Benzene concentrations were reduced between 1982 and 1988 from 17 mg/L to 0.6 mg/L, the highest concentration still residing at the point of origin and low level residues (10 µg/L) having migrated as far as 54M. (See Figs. 18a and 19a.)

This reduction is largely attributable to the experimental remedial operations and dispersion. From evidence gathered in a microcosm study (Wilson *et al.* 1986) and in an anoxic aquifer (Barker *et al.* 1987), biodegradation was likely to be slow with benzene as the most refractory compound.

Table 8. Volatile Organic Compounds Present in the Outwash Aquifer, May 1988

Compound	Concentration (µg/L)	Frequency[a] %	Aqueous solubility[b] (mg/L)	% of aqueous solubility[c]
Dioxane	~ 300-2000	13	∞ [d]	
Diethyl ether	<2-658	68	84300(15°C)[e]	0.0008
Tetrahydrofuran	20-230	32	∞ [d]	
Benzene	0.9-646	65	1780(20°C)[e]	0.04
Toluene	<0.5-2.5	30	470(16°C)[e]	0.0005
Ethylbenzene	2	3	140(15°C)[e]	0.001
o-Xylene	10	3	175(20°C)[e]	0.006
2-Ethyltoluene	2.1	3	n.a.	
Dichloromethane	4-60	16	15938	0.0004
Trichloromethane	<2-164	51	8983	0.002
Tetrachloromethane	<2-7.5	11	683	0.001
Chloroethane	<6	3	5742	0.0001
Chloroethene	<1-40	13	2763(25°C)	0.0015
1,1-Dichloroethane	<1-614	54	5193	0.012
1,1,-Dichloroethene	0.9-60	43	6650	0.0009
1,2-Dichloroethane	3.9-58	30	8579	0.0007
Trans-1,2-dichloroethene	<1-2.8	16	6260(25°C)	0.00004
Cis-1,2-dichloroethene	<1-7.5	19	4000	0.0002
1,1,1-Trichloroethane	<3-520	43	1707	0.03
1,1,2-Trichloroethane	<2	3	4564	0.00004
Trichloroethene	7.4-583	41	1061	0.055
Tetrachloroethane	5-15	8	763	0.002
Tetrachloroethene	2-105	27	149	0.07
1,2-Dichloropropane	3.2-11	5	2300	0.0005
Chlorobenzene	1.1-134	11	193	0.069
1,1,2-Trichloro-1,2,2-trifluoroethane	<5-2725	35	136	2

[a] Frequency of identification of compounds in 37 samples collected from the network shown in Fig. 3.

[b] Solubilities from Horvath (1982) at 10°C unless otherwise indicated.

[c] Represented by the maximum measured concentration.

[d] Miscible.

[e] Solubility from Verschueren (1983).

n.a. = No data available.

Table 9. Volatile Organic Compounds Tentatively Identified by Computerized Library Searches

1,2-Dichloro-1,2,2-trifluoroethane
1,1,2-Trichloro-1,2-difluoroethane
Dichlorodifluoromethane
Chlorotrifluoroethene
Diisopropyl ether
Pentane
Methoxybenzene
Diethyl sulfide
Dimethyl sulfide
Methylethyl sulfide
Tetrahydrothiophene
Chloropropene
1,2-Dibromomethene

All concentrations estimated to be <50 µg/L.

Table 10. Volatile Organic Compounds (in µg/L) Identified in the Monitoring Well Network, May 1988

Sample	124P	125P	126P	127P	128P
Dioxane				~330	
Diethyl ether	643	72	331	122	
Tetrahydrofuran	70		230	180	
Benzene	2.8	120			
Toluene		3			
Ethylbenzene		3			
m+p-Xylene		2.6			
o-Xylene		10			
Trichloromethane		52			<2
Tetrachloromethane					<4
Chloroethane	<1				
1,1-Dichloroethane	2.6	24			0.9
1,2-Dichloroethane	10.3	14			
1,1-Dichloroethene		155			<2
Trans-1,2-dichloroethene					<1
Cis-1,2-Dichloroethene					<1
1,1,1-Trichloroethane		193			<3
Trichloroethene		505			
Tetrachloroethene		60			
1,2-Dichloropropane		12			
Chlorobenzene		68			
1,1,2-Trichloro-1,2,2-trifluoroethane		200			

Blank spaces indicate compound not detected.

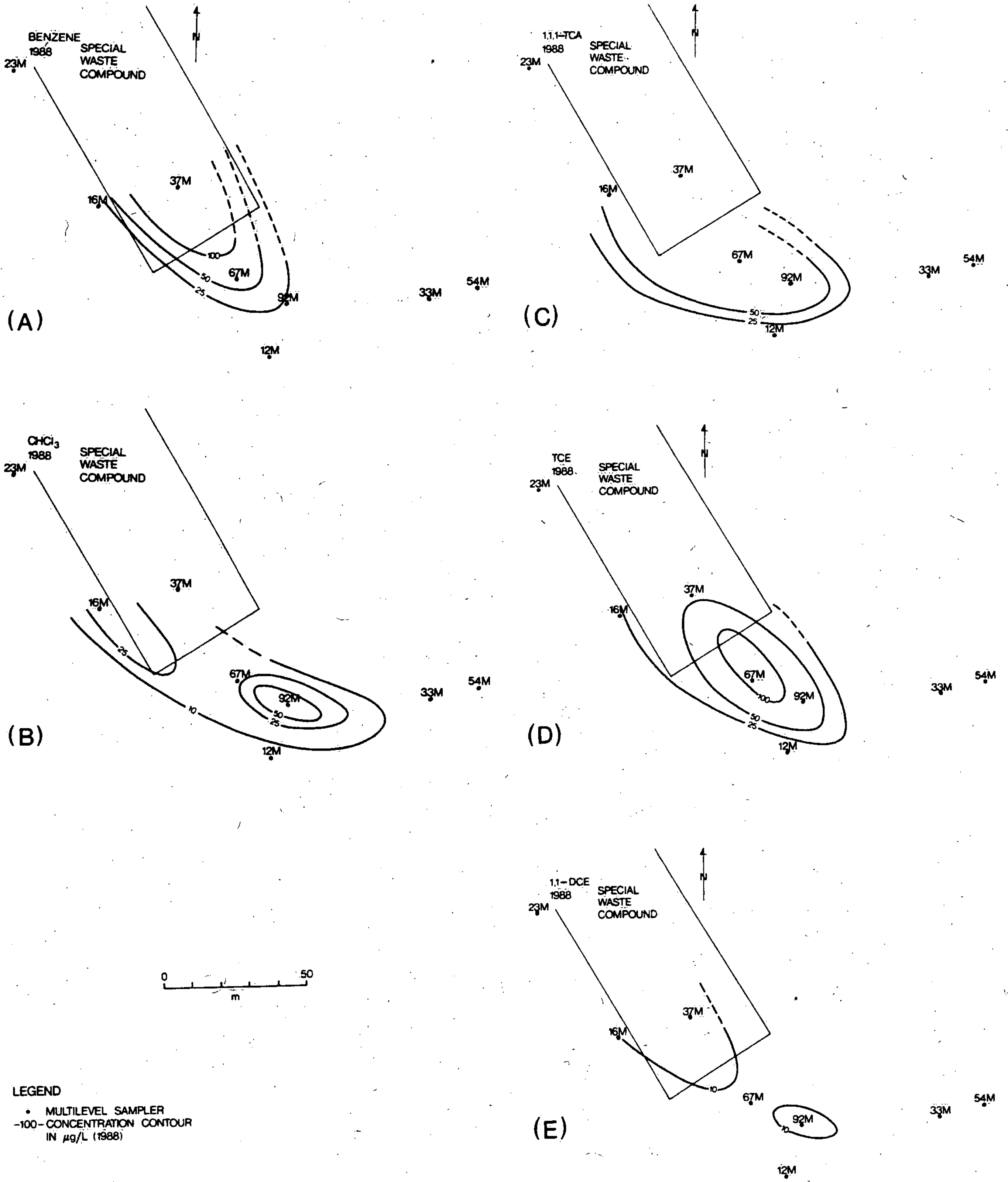

Figure 18. Concentration contours (in $\mu g/L$) of (a) benzene, (b) chloroform, (c) 1,1,1-trichloroethane, (d) trichloroethene, and (e) 1,1-dichloroethene in the immediate proximity of the SWC, May 1988.

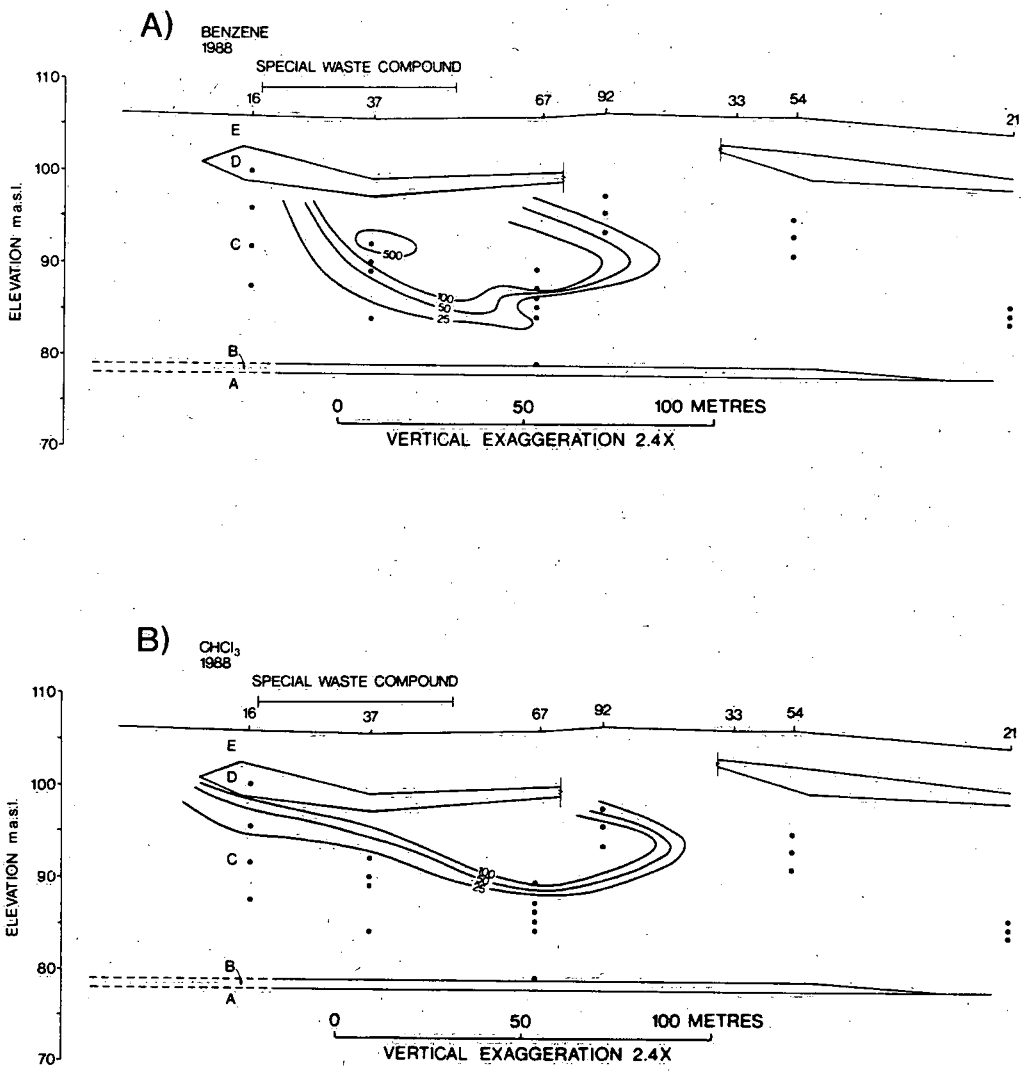

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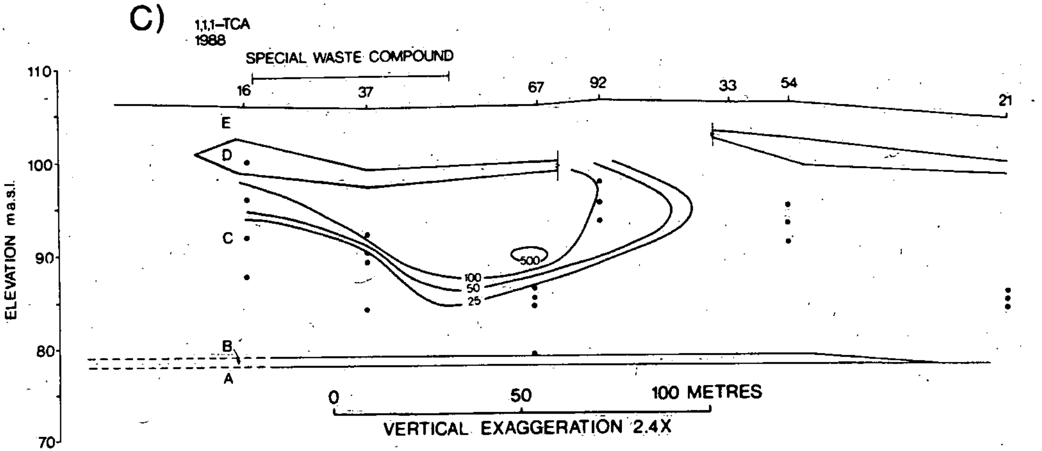
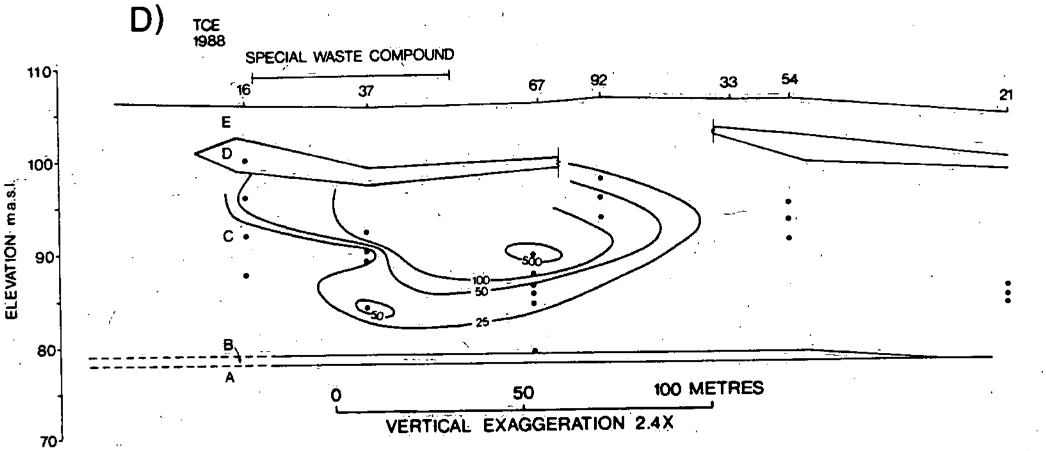
Figure 19. Concentration contours (in µg/L) of (a) benzene, (b) chloroform, (c) 1,1,1-trichloroethane, (d) trichloroethene, and (e) 1,1-dichloroethene along the cross section shown in Figure 16, May 1988.

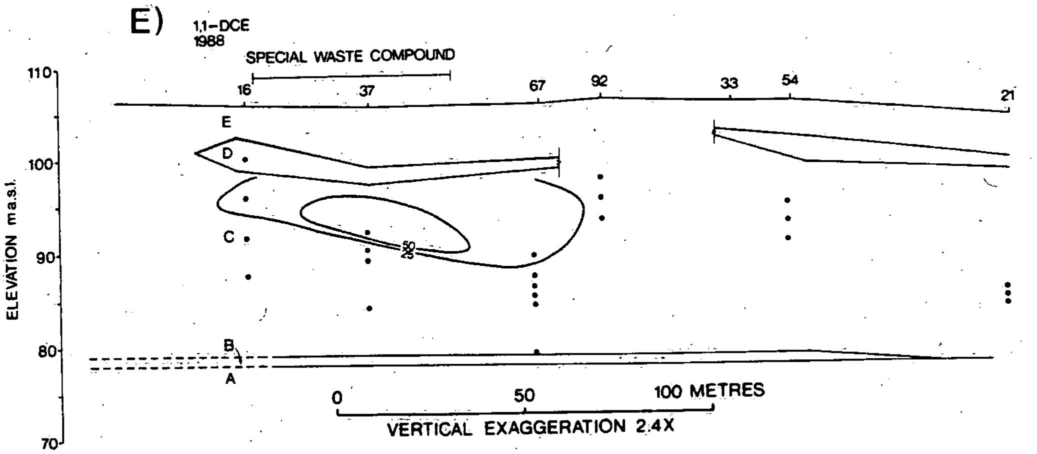
C)


1,1,1-TCA
1988
SPECIAL WASTE COMPOUND
16
37
67
92
33
54
21
E
D
C
B
A
500
100
50
25
ELEVATION m.a.s.l.
110
100
90
80
70
0
50
100 METRES
VERTICAL EXAGGERATION 2.4X

D)


TCE
1988
SPECIAL WASTE COMPOUND
16
37
67
92
33
54
21
E
D
C
B
A
500
100
50
25
50
ELEVATION m.a.s.l.
110
100
90
80
70
0
50
100 METRES
VERTICAL EXAGGERATION 2.4X

E)


1,1-DCE
1988
SPECIAL WASTE COMPOUND
16
37
67
92
33
54
21
E
D
C
B
A
50
25
ELEVATION m.a.s.l.
110
100
90
80
70
0
50
100 METRES
VERTICAL EXAGGERATION 2.4X

Chlorinated Aliphatics

Plumes for these compounds as found in May 1988 are depicted in Figures 18b-e and 19b-e. The centres of mass for trichloromethane ($CHCl_3$), 1,1,1-trichloroethane (TCA), and trichloroethene (TCE) occur at multilevel 67M, although that of the degradation product 1,1-dichloroethene (1,1-DCE) occurs at 37M-9.

Degradation of tetrachloroethene (PCE) and TCA appears to have occurred because of the presence of several metabolic intermediates such as 1,1-dichloroethane (1,1-DCA), chloroethene (VC), cis- and trans-1,2-dichloroethene (CIS and TRANS), and 1,1-DCE (Tables 8 and 10, Fig. 20) in the same or in adjacent wells. Most samples contained both TCA and TCE and mixtures of their respective metabolites such that it is difficult to ascertain which pathway predominates, since some metabolites such as 1,1-DCE and VC are common to both. Possible inferences from spatial distribution are obscured because of the physical disturbances of the ground water. By integrating the concentration over the wells that have shown greatest contamination over the time period and considering the product distribution, it is possible to evaluate the results of biotransformation without having to consider differences in analytical methodologies, absolute concentrations, or the transport component. The product distributions for 1983 and 1988 are listed in Table 11.

From these data, it is possible to infer that degradation was indeed occurring since lesser chlorinated compounds increased relative to the more highly chlorinated compounds. Reductive dehalogenation reactions were prevalent, but elimination may have occurred to form 1,1-DCE from TCA, although this cannot be proven in the case of the chlorinated alkanes, because 1,1-DCE may have been formed from the reductive dehalogenation of TCE. Some of the products observed for chlorofluorocarbon degradation support the existence of this pathway (vide infra). The observed product ratio of CIS and TRANS was different from what has been reported previously (Wood *et al.* 1985; Parsons *et al.* 1984). In most samples, the two products were present in equivalent amounts, whereas these authors quoted a ratio of 25 : 1 for CIS : TRANS. CIS was found to be rapidly degraded in a microcosm study of methanogenic aquifer material (Wilson *et al.* 1986), hence it would not be expected to be present in higher concentration. One of the possible explanations for this difference is analytical; indeed, some methods did not allow for good separation of the two isomers.

Chlorofluorocarbons

The presence of 1,1,2-trichloro-1,2,2-trifluoroethane (CFC113) had not been reported initially, probably because it is one of the first compounds to elute during gas chromatography. Consequently, its chromatography requires cryotrapping, which had not been used in the earlier analyses, i.e., before 1988. Its presence in a laboratory waste disposal site such as Gloucester is not surprising because this solvent is used in oil and grease determinations. Its mobility would be expected to be similar to that of PCE because their octanol-water partition coefficients are similar (log K_{ow} 2.57 for CFC113 vs 2.60 for PCE; Roy

Table 11. Product Distributions Expressed as Percentage of Total Chlorinated Ethanes/Ethenes Found in 37M, 67M, and 92M

Compound	1983	1988
TCA	27.3	27.6
1,1-DCA	39.5	25.4
1,1-DCE	1.4	5.1
PCE	9.7	3.3
TCE	22.0	36.8
CIS	n.m.	0.5
TRANS	0.1	0.2
VC	n.m.	1.0

n.m. = Not measured.

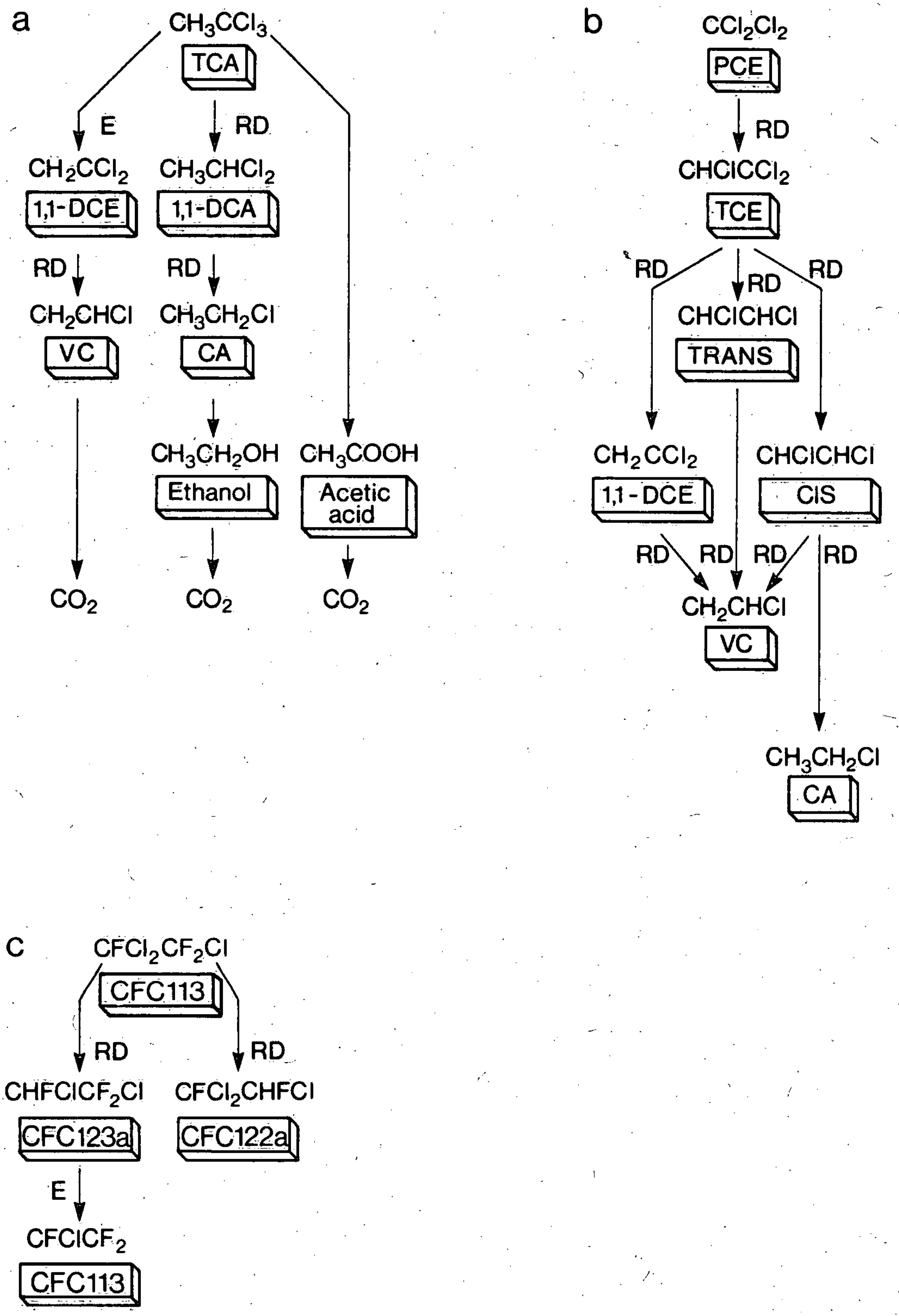


Figure 20. Proposed anoxic biotransformation pathways for (a) 1,1,1-trichloroethane or TCA under methanogenic conditions (Vogel and McCarty 1987), (b) tetrachloroethene or PCE (Barrio-Lage et al. 1986), and (c) 1,1,2-trichloro-1,2,2-trifluoroethane or CFC113 (Lesage et al. 1990). RD = Reductive dehalogenation reaction, E = Elimination or dehydrodehalogenation reaction. See text for other abbreviations.

and Griffin 1985), and indeed it had migrated to the same area. It seems much more refractory to biological transformation and is now the compound found in largest concentration at the site (2.7 mg/L in 67M-11), more than five times the TCE and PCE concentrations. Unfortunately, no historical disposal data is available for CFC113, which means that its attenuation over time cannot be assessed.

1,2-dichloro-1,2,2-trifluoroethane (CFC123a) was also found in the same samples, but at much lower concentrations (10-50 µg/L) than CFC113. CFC123a could conceivably have been present as an impurity in CFC113. According to the manufacturer (S. Lauridsen, August 1988, Dupont Canada, Maitland, Ont., pers. comm.), it can comprise up to 0.07% of the product in the synthesis reactor, prior to initial purification. The concentration of CFC123a found in the samples comprised from 0.5 to 50% of the CFC113 concentration, far in excess of the possible impurity level. Furthermore, a sample of the technical products as sold to redistillers was analyzed and found not to contain any detectable CFC123a.

CFC123a could also have been formed by pyrolysis. Indeed CFC123a has been identified in the combustion products of CFC113 in the presence of steel, copper, and lubricating oil (Srinivasan et al. 1984). The authors did not confirm it as a product of pyrolysis, referring to the fact that it could be found as an impurity in the technical product. The major product of combustion they observed was chlorotrifluoroethene (CFC1113), which was also identified in multilevel 67M-11 at an estimated concentration of <10 µg/L. Therefore, the observed product ratio (CFC123a: CFC1113) was the reverse of what would be expected from combustion of CFC113 alone.

CFC123a could also have been the product of in-situ microbial degradation. In support of this hypothesis was the fact that in spite of their predicted higher mobility, CFC123a and CFC1113 were found in the same multilevels as CFC113, and their ratio was not consistent with the known pyrolysis products. Furthermore, these compounds were found in the midst of an anoxic zone where there was evidence of sulfate reduction. The formation of CFC1113 from CFC113 might follow the same metabolic pathway as the formation of VC from TCA (Fig. 20a). The first reaction is one of reductive dehalogenation, followed by the elimination of HCl to form the olefin. The mechanism of the reaction has been postulated as involving free-radical formation involving cytochrome P-450 in the case of the metabolism of 2-bromo-2-chloro-1,1,1-trifluoroethane (Monig and Asmus 1984). We also tentatively identified 1,1,2-trichloro-1,2-difluoroethane (CFC122a) which, if it was formed from CFC113, would be most likely to arise via a free-radical mechanism, which would tend to be less specific than a substitution mechanism, in which the loss of F over Cl is not favored.

CFC113 has been gradually replacing PCE as a dry-cleaning solvent and is widely used in the electronic industry, where it has caused substantial ground-water contamination (Esau and Chesterman 1988). This work shows that it may be even more of a threat to ground water, because it appears more persistent than PCE. It was found to act as a co-carcinogen with polyaromatic hydrocarbons without any inference to the possible pathway (Mahurin and Bernstein 1988). Both CFC113 and CFC123a were found to be nonmutagenic in Ames tests; however, the authors caution this may not be a valid indicator of mammalian toxicity. This is because these compounds could be analogous to 1-fluoro-2,2,2-trifluoroethane which, although it was found to be nonmutagenic in Ames test, was found to be carcinogenic in an animal study (Longstaff *et al.* 1984). CFC1113 is used in its gaseous form in the polymer industry and is classified as highly toxic (268 mg/kg oral; Sax 1984); its presence in ground water is therefore of concern.

Sulfur Containing Compounds

As noted above, that part of the outwash aquifer associated with the chloroalkane/ene plume is a sulfide redox zone and the presence of organo-sulfur compounds was suspected from their characteristic odour. The organo-sulfur compounds identified (Table 9) were dimethyl sulfide, diethyl sulfide, methyl ethyl sulfide, and tetrahydrothiophene in monitors 37M, 67M, and 125P.

Because these were found in areas where chlorinated organic solvents were present in high concentrations, the possibility of reaction between the two species was considered, but no chlorinated organosulfur products were identified. Indeed, although sulfides may react with halogenated aliphatics to form mercaptans (Schwarzenbach *et al.* 1985), and, although sulfur nucleophiles are more reactive than their oxygen analogs, their relative concentrations in aqueous systems are so much smaller that these reactions are seldom observed (Vogel *et al.* 1987). The formation of chloroalkylsulfones, which are resistant to biodegradation, has been reported (Headley 1987), but their presence has been attributed to discharge from biologically treated kraft mill effluents, which means that their occurrence would constitute an exception rather than the norm in ground-water systems.

DISCUSSION OF CONTAMINANT TRANSPORT MECHANISMS

The widespread migration of chloride and organic chemicals shown in Figures 9, 10, 11, 18, and 19 is due to solute transport by ground water. Jackson *et al.* (1985) showed that this solute transport was affected by sorption of the organic chemicals by aquifer materials ($f_{oc} = 0.06\%$). Due to hydrophobic processes, sorption causes the organic solutes to migrate at differing mean velocities, which are functions of the octanol-water partition coefficient (K_{ow}) of the compound. Solutes with K_{ow} values that are relatively small, such as the aliphatic ethers (i.e., 1,4-dioxane, diethyl ether, and tetrahydrofuran), have travelled the farthest. For example, large concentrations of oxygenated solvents are present at 126P and 127P, but aromatic hydrocarbons and chlorinated aliphatics have been retarded in their transport and are to be found mainly beneath the SWC (Figs. 18 and 19). Solute transport and sorption produce retardation factors (ratio of ground-water velocity to contaminant velocity) that vary from 1.6 (1,4-dioxane) to >20 (carbon tetrachloride).

In addition to aqueous-phase solute transport, there is the possibility that organic contaminants may migrate through the unsaturated and saturated zones of the ground-water flow system as separate phases whose behaviour is a function of their relative saturation, density, and viscosity. Of particular concern are those organics that constitute a threat to human health, such as the halogenated aliphatics. These may be present in the flow system as either pools of liquid chemical ("free product") in contact with some relatively impermeable hydrostratigraphic unit (e.g., bedrock or silt) or as ganglia (residual droplets) left behind in pores as the DNAPLS sank through the aquifer system (Schwille 1984; Sitar *et al.* 1987; Schwille 1988; Feenstra and Cherry 1988).

The occurrence of DNAPLs in aquifer systems is particularly hard to verify unless the fouling of downhole equipment or the saturation of cores is observed. However, these instances would be the exceptions rather than the rule, due to the relatively small size of most pools or ganglia and the relatively large distances between boreholes. Typically, DNAPLs produce concentrations in nearby monitoring wells that are less than 10% of their aqueous solubility (Feenstra and Cherry 1988). Concentrations approaching the solubility limit are not measured in ground waters due to the heterogeneous spatial distribution of ganglia or pools in the subsurface, mass transfer constraints on their dissolution, and the dilution of the dissolved components during ground-water transport, transformation and sampling (Feenstra and Cherry 1988; Anderson *et al.* 1987; Hunt *et al.* 1988a).

The likelihood of pools of DNAPL being present in the outwash aquifer immediately beneath the SWC would appear remote. Figures 21 and 22 show vertical profiles for three DNAPLs and benzene at multilevel samplers within and close to the SWC in 1982, 1984, and 1988. The chlorinated aliphatics show no persistent or monotonic increase with depth as might be expected if a DNAPL pool had sunk vertically to the bottom of the aquifer. The relatively large concentrations of volatile organic chemicals measured in 1982 and 1984 are not observable in the 1988 profile, due to diluting effects of the experimental pump and treat operations and of natural ground-water dispersion and biotransformation. Furthermore, a bedrock monitoring well (37P-3, see Fig. 16), located beneath the SWC and hydraulically connected to the outwash aquifer, remains uncontaminated. Had a DNAPL pool been present, the termination of experimental pumping in 1986 would have been followed by an increase in the concentration of contaminants dissolved in ground water.

However, this conclusion may not apply to those parts of the outwash aquifer adjacent to the SWC that are not confined by the aquitard (Unit D). The samples from the 1988 survey of the SWC, which were analyzed by GC-MSD with cryotrapping, identified the presence of large quantities of the chlorofluorocarbon, CFC113, and its possible transformation products. At multilevel sampling point 67M-11, some 20 m south of the SWC, the concentration of CFC113 was approximately 2% of its aqueous solubility at 10°C, a percentage indicative of the possible presence of DNAPLs. Furthermore, concentrations of other organic residues have exceeded 1% of their solubility as is shown by chloroform and PCE in Figure 23.

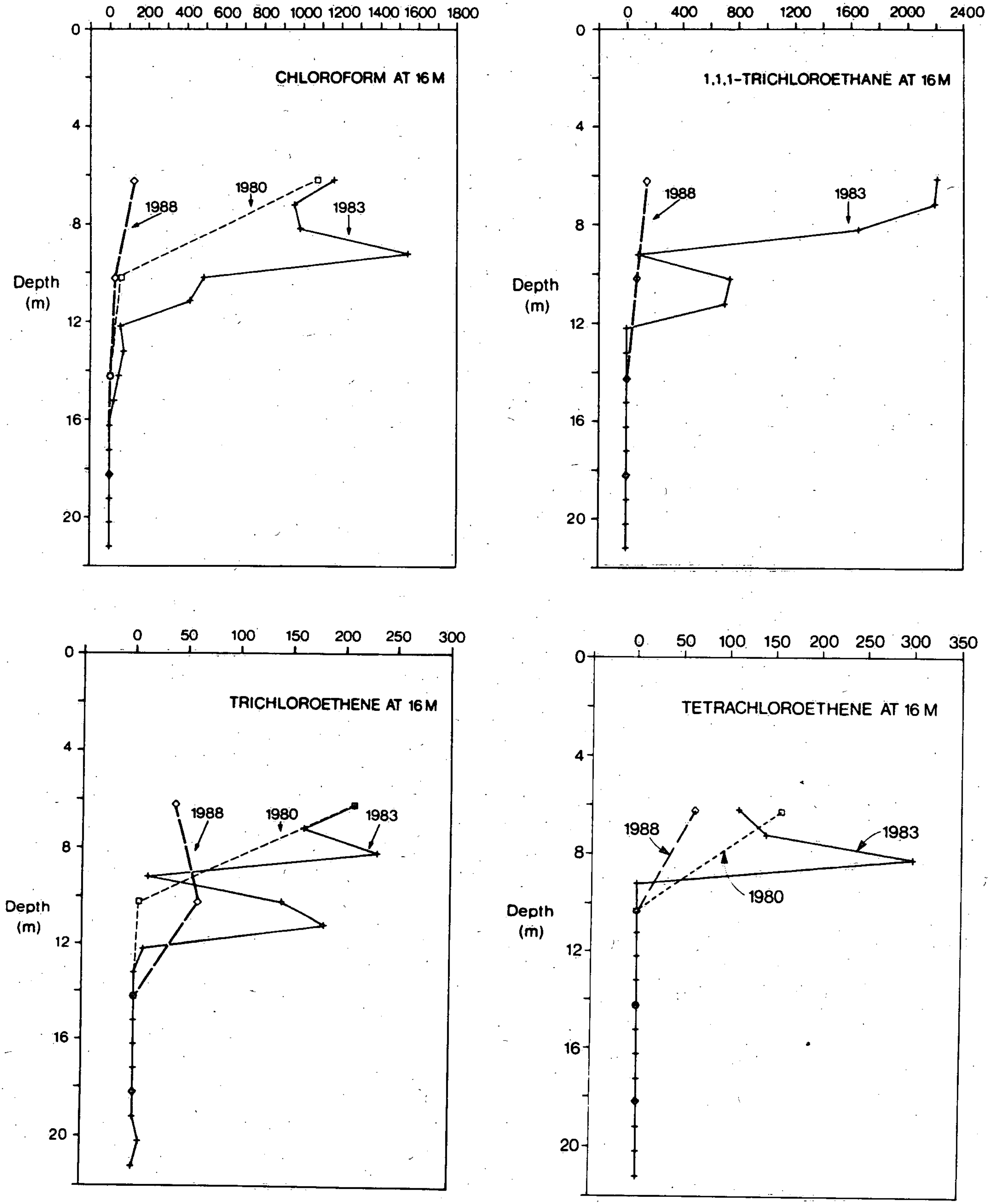


Figure 21. Vertical concentration profiles (in $\mu g/L$) of four organic solvent residues at monitor 16M during 1980, 1983, and 1988.

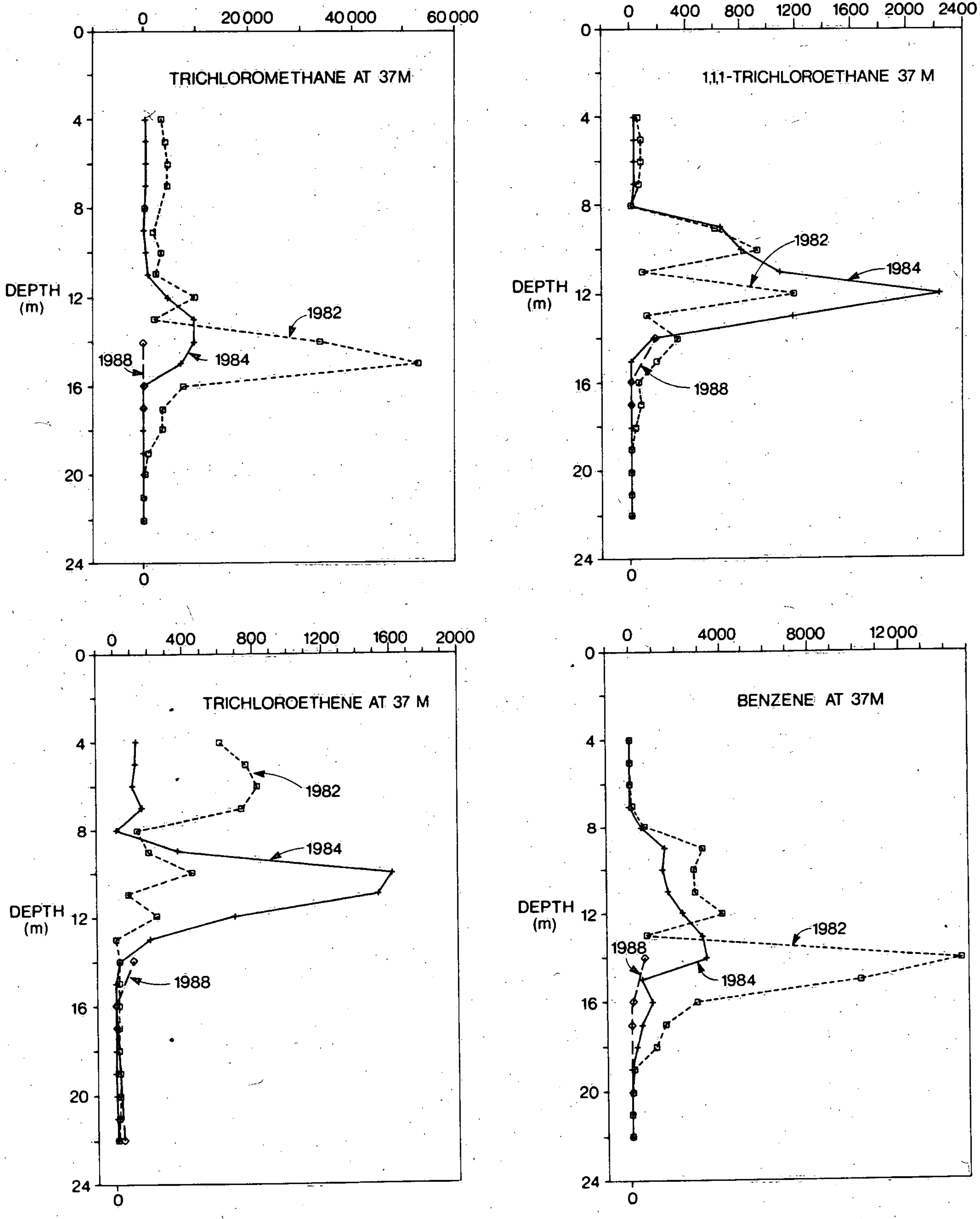


Figure 22. Vertical concentration profiles (in µg/L) of four organic solvent residues at monitor 37M during 1982, 1984, and 1988.

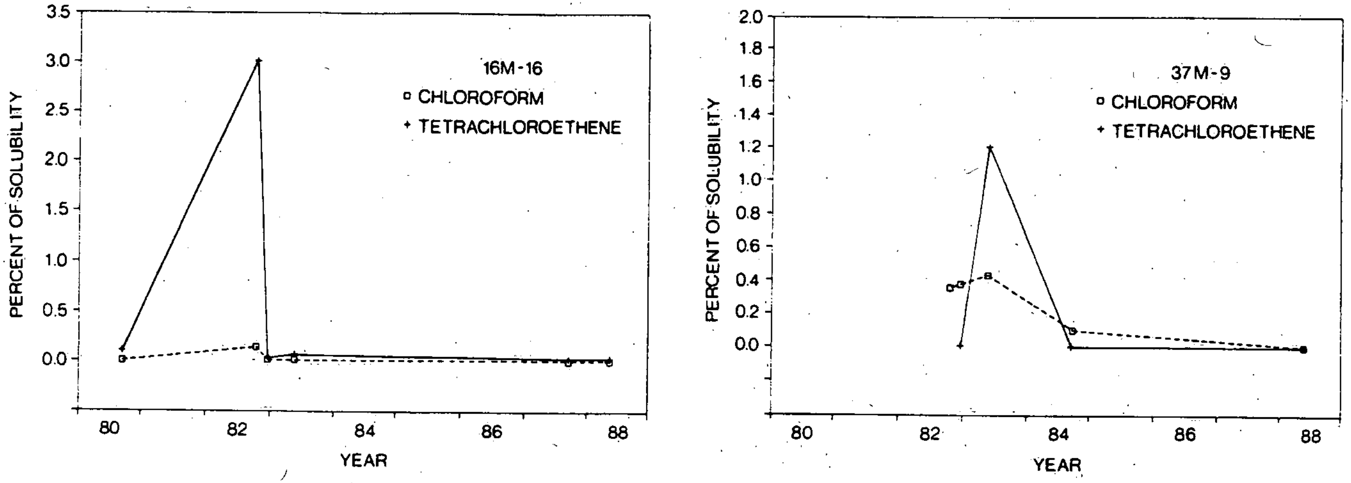


Figure 23. Time series data from two multilevel monitoring points within (37M) or adjacent to (16M) the SWC showing that concentrations of volatile organic chemicals have always been a small percentage of the solubility of any compound present in the ground water.

This analysis suggests that, following disposals of CFC113, the chlorofluorocarbon may have migrated as viscous fingers (Schwille 1988; Anderson *et al.* 1987) through the water-table aquifer (Unit E) and, by some pathway, through or around the aquitard (Unit D) and into the outwash aquifer (Unit C). A thin, clayey gravel layer was identified in drilling logs at an elevation immediately below sampling point 67M-11. This indicates that a small pool or extended ganglia may well exist at this location and be sitting on a low permeability strata quite common to this type of glaciofluvial sediment. This monitor also contains the maximum concentrations of other volatile organic chemicals (1,2-dichloroethane, chlorobenzene, TCA, TCE, and PCE) observed at the site; these may have been part of the same disposal. This scenario implies that ganglia or pools exist along the pathways taken by the viscous fingers, particularly at the contact between the water-table aquifer and the aquitard.

Mobility of Organic Priority Pollutants in the Outwash Aquifer

The migration of the organic priority pollutants discussed in Chapters 2 and 3 is mainly the result of solute transport due to ground-water flow through the outwash aquifer as well as solute sorption, resulting in "chromatographic dispersion" (Patterson *et al.* 1985). In order to predict the rates of migration of the organic solutes, two types of experiments have been conducted to determine the velocity of the organic solutes relative to that of a conservative or nonreactive tracer, e.g., chloride or tritium.

The first of these experiments was a field test within the outwash aquifer, known as the "purge well test", conducted to test the efficacy of using a pump and treat system to decontaminate the outwash aquifer. The second type of experiment involves the use of laboratory columns of Gloucester aquifer sands to estimate the migration rates. The parameter of interest in both experiments is known as the retardation factor, R_f, which is defined by Freeze and Cherry (1979, p. 404) as

$$R_f = 1 + (\rho_b K_D / n)$$

where ρ_b is the bulk density, K_D is the solute distribution (or partition) coefficient and n is the porosity of the aquifer sediments. The distribution coefficient is merely the slope of a linear adsorption isotherm for the organic solute and aquifer sediment under consideration. In the case of a nonreactive solute, $K_D = 0$ and $R_f = 1$.

Thus the retardation factor allows one to estimate the mobility of a particular organic solute in a particular aquifer material from the following expression:

$$V_C = V_{GW}/R_f$$

where V_C and V_{GW} are the mean velocities of the contaminant solute and the ground water, respectively. This information is crucial in predicting the farthest extent of contaminant migration and in assessing the duration of aquifer remediation.

PURGE WELL TEST

In the fall of 1983 a test was undertaken by Whiffin and Bahr (1985) to study the decontamination of the outwash aquifer using purge wells. The site chosen for this experiment was in the middle of the chloride plume, i.e., near multilevel 54 (see Fig. 6). The test was conducted by injecting uncontaminated, tracer-labelled ground water into a fully penetrating well at the rate of about 140 L/min for the first day, about 95 L/min for the following 4 days and 140 L/min for the sixth and final day, while withdrawing at the same rates from a second, similar well 5 m away (see the inset in Fig. 24). Between the two wells was situated a multilevel sampler, M1, from which samples were taken throughout the 6-day period of the test. These were analyzed for two injected tracers (iodide and a fluorinated benzoic acid), both presumed to be nonreactive, and for three oxygenated solvents present in that part of the aquifer — 1,4-dioxane, tetrahydrofuran and diethyl ether.

Retardation factors (R_f) for each of the contaminants were estimated by dividing the $C/C_0 = 0.5$ time for the disappearance of the chemical due to its withdrawal or "purging" by the $C/C_0 = 0.5$ time for the arrival of the nonreactive tracers at the multilevel. These R_f values are shown in Table 12 along with values determined independently from (1) plume mapping (Patterson et al. 1985) and (2) the equation of Schwarzenbach and Westall (1981) and the retardation equation. The close agreement between the R_{f1} and R_{f2} values suggest that the plume-length retardation factors adequately describe the relative mobility of the organic contaminants.

The differences between the R_f value obtained by the equation of Schwarzenbach and Westall (i.e., R_{f4}) with the other values indicate that this predictive equation will yield results that are increasingly in error as compounds with larger K_{ow} values are considered. This is likely due to sorption of the organic solutes on mineral surfaces, a factor not incorporated into the prediction equation (Curtis *et al.* 1986).

The behaviour of the organic contaminants during desorption is shown in Figure 24, in which the time axis is replaced by the number of pore volumes pumped through the test section of the aquifer (a pore volume is defined by Whiffin and Bahr (1985) as "the volume of water required per unit aquifer thickness to displace the interstitial ground waters located between the injection and monitoring points as defined by the breakthrough of the 50% concentration level for the nonreactive tracer"). While dioxane seems to be purged rapidly from the test section within the aquifer, aquifer, tetrahydrofuran (THF) and diethyl ether (DEE) require longer purge times and display pronounced tailing, which is probably due to nonequilibrium sorption/desorption reactions.

This tailing is consistent with that expected when the transport of these contaminants is influenced by a linear reversible sorption/desorption process that does not proceed at a rate sufficient to reach local equilibrium during desorption at these flow velocities (3 m/d at M1 or roughly 50 times the natural ground-water flow velocity). THF also exhibits an abrupt increase in concentration following the

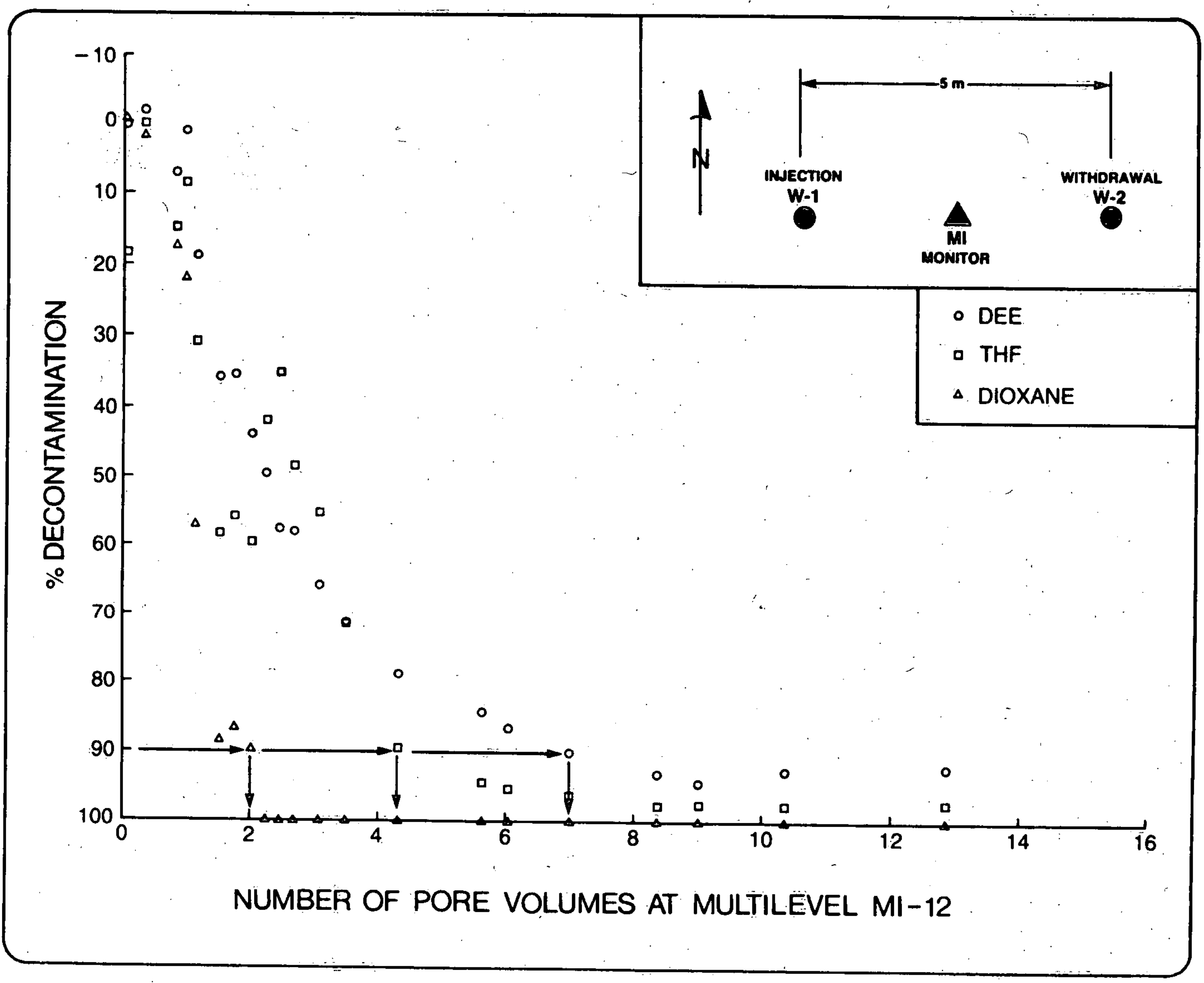


Figure 24. Rate of reduction of contaminant concentrations (diethyl ether or DEE, tetrahydrofuran or THF, and 1,4-dioxane) during the purge well test.

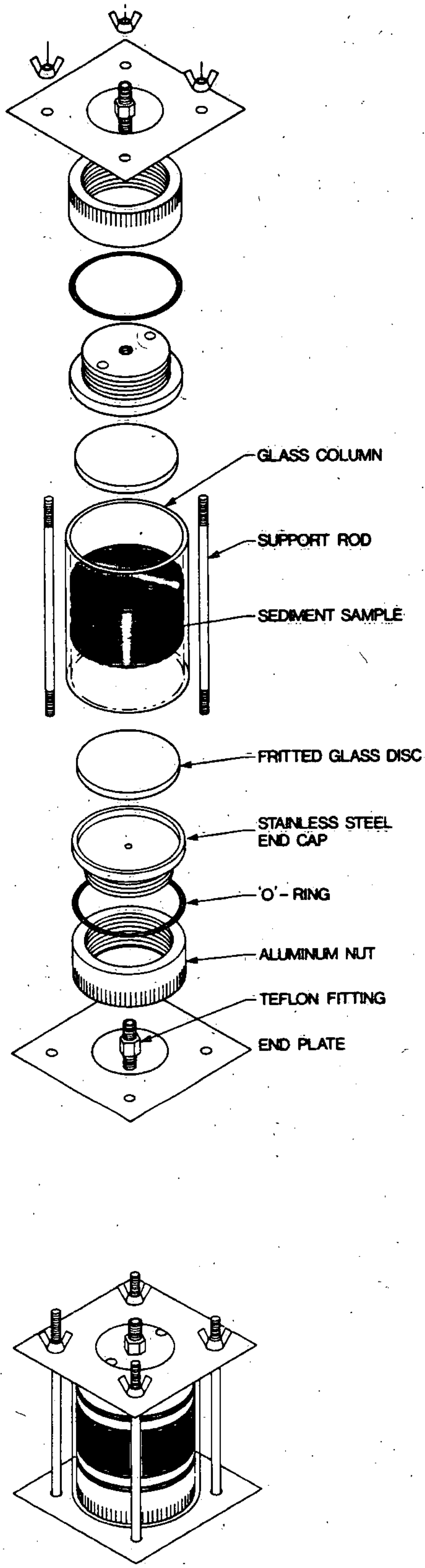


Figure 25. Exploded view of a laboratory soil column.

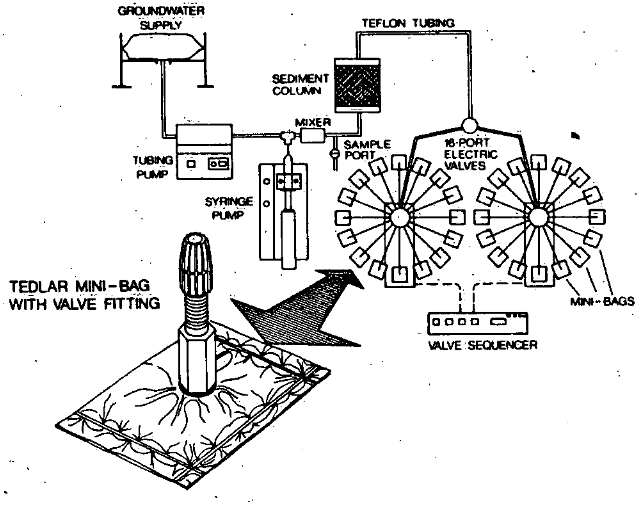


Figure 26. Simplified diagram of the complete column apparatus.

decrease in flow rate after 24 h. The rate limiting process may be either the sorption mechanism itself or a physical diffusion process (Bahr 1989).

LABORATORY COLUMN TESTS

The laboratory soil column offers the most practical and inexpensive method of estimating R_f in sand and gravel aquifers, e.g., Reynolds *et al.* 1982; Lee *et al.* 1988a. The laboratory will need access to a variety of variable flow rate pumps, an effluent fraction collecting system and some means for analyzing the effluent, e.g., GC or liquid scintillation counter. In estimating the retardation factor of a volatile organic chemical (VOC), such as diethyl ether or tetrachloroethene, the VOC is treated as a tracer and injected into ground water flowing into the column.

The method used to estimate R_f values for VOCs in the Gloucester aquifer involves packing a short glass column (10-30 cm long) with aquifer material to the bulk density of the aquifer. The column, shown in Figure 25, is fitted with end plates to allow the entry and elution of the tracer and ground water. It is then filled slowly from below with de-aerated ground water so that air bubbles are not trapped inside it.

Figure 26 shows the complete soil column apparatus, which is maintained at 10°C in a cold room. The ground-water supply was taken from multilevel 84M-6, upgradient of the SWC, and was passed through a 0.2-μm filter. It is then pushed into the column via a tubing pump, which regulates the velocity of the ground water. Before its entry into the column, the tracers (i.e., organic contaminants and iodide) are added via a syringe pump and this solution is passed through a mixer containing stainless steel balls. The input concentration, C_0, is measured by taking a sample in a Tedlar bag from a port immediately before the column. The effluent from the column is discharged into evacuated Tedlar bags, each fitted with a Teflon on/off valve. The filling of the bags is controlled by a valve sequencer. The bags are disconnected after filling and a syringe is used to remove a sample of the effluent for injection into a purge and trap GC/MSD. The plot of the normalized effluent concentration (C/C_0) versus time is known as the breakthrough curve or BTC.

The retardation factor can be estimated by several methods. The simplest of these is to compare the arrival times of the organic tracer and that of a non-reactive tracer, such as iodide, using the values at $C/C_0 = 0.5$ (or $C/C_{max} = 0.5$ if C_0 is not reached). However, this method ignores the physical and chemical nonequilibrium processes that cause substantial tailing and asymmetry in the BTC (e.g., Lee *et al.* 1988a). As Bahr (1989) has shown, these processes are important in pump and treat operations and should be incorporated in the measured value of R_f.

Parker and van Genuchten (1984) have developed a suite of solute transport models to fit to BTCs using optimization methods to determine R_f and other parameters. Two of the models are of interest in the present circumstances. The first fits a linear equilibrium assumption (LEA) model to the BTC and thus assumes that sorption of the organic tracer is instantaneous, reversible, and follows a linear adsorption isotherm. The second, a nonequilibrium model, treats the sorption reaction as involving two different types of sites or regions in the aquifer material: an LEA fraction and a second fraction for which sorption is considered to be time dependent or rate limited.

Table 12. Retardation Factors

Chemical	log K_{ow}	R_{f1}	R_{f2}	R_{f3}	R_{f4}	R_{f5}	R_{f6}	R_{f7}
Dioxane	-0.27	1.6	1.4	1.6	1.0	1.1	1.2	1.4(2)
THF	0.46	2.2	2.2	2.5	1.0			
DEE	0.89	3.3	3.0	3.4	1.1	1.7	1.6	2.2(2) 3.2(4)
12DCA	1.48	7.6		5.7	1.2	7.2	5.0	13.1(4)
TCM	1.97			9.3	1.3	7.5	5.0	14.9(4)
Benzene	2.04	8.8		10.0	1.4	14.3	8.0	30.0(4)
11DCE	2.13			11.0	1.5	10.7	7.0	18.8(4)

Retardation factors based on

1. Field data (length of organic plume vs length of chloride plume)
2. Purge well evaluation test
3. Correlation $\log (R_f-1) = 0.5*\log K_{ow} - 0.065$ derived from field data (Patterson et al. 1985)
4. Equation $R_f = 1+(\text{bulk density}/\text{porosity})*K_d$ where $K_d = 3.2*f_{oc}*(K_{ow})*\exp 0.72$ (Schwarzenbach and Westall 1981)
5. Column test #3 (centre of mass of organic vs centre of mass of iodide)
6. Column test #3 (C/Cmax = 0.5 of organic vs C/Cmax = 0.5 of iodide)
7. Column test #3 (CXTFIT: number in parenthesis refers to Model 2 (LEA) or 4 (nonequilibrium))

Notes:

Bulk density - 1.76 g cm^{-3}
Porosity - 0.32
f_{oc} - 0.00075

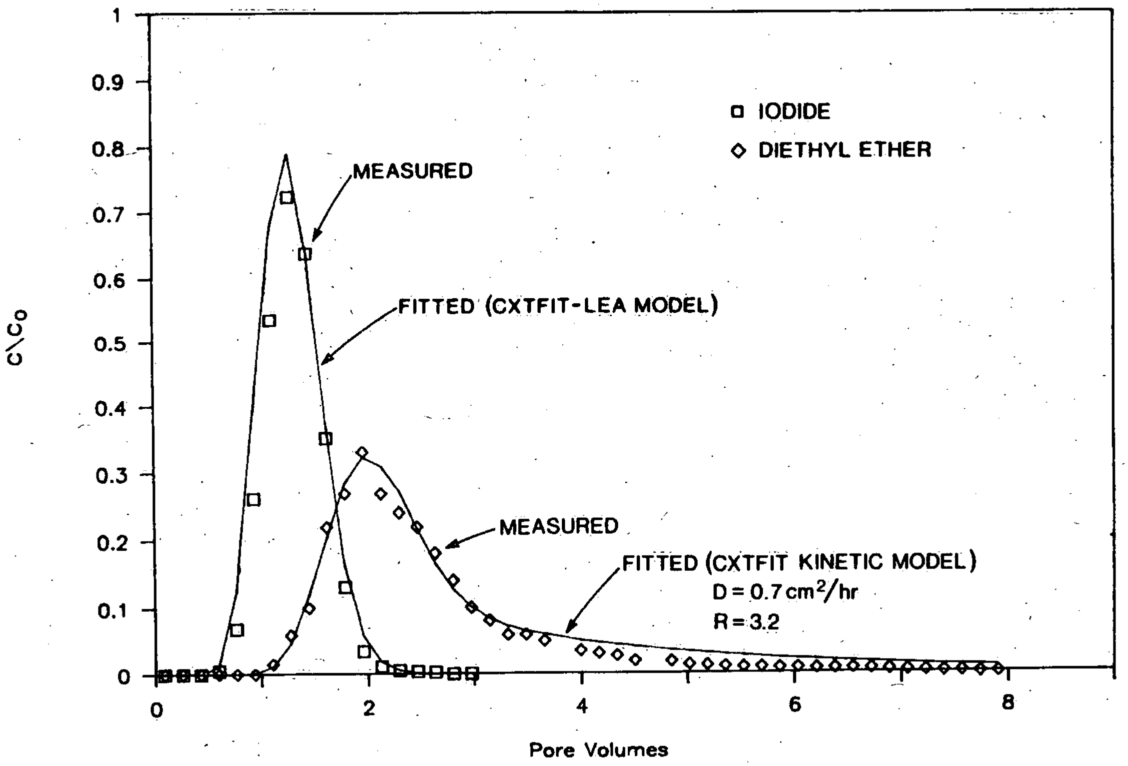


Figure 27. Breakthrough curves of iodide (V = 45 cm/d) and diethyl ether showing both measured and fitted values using the LEA model (iodide) and the kinetic model (diethyl ether).

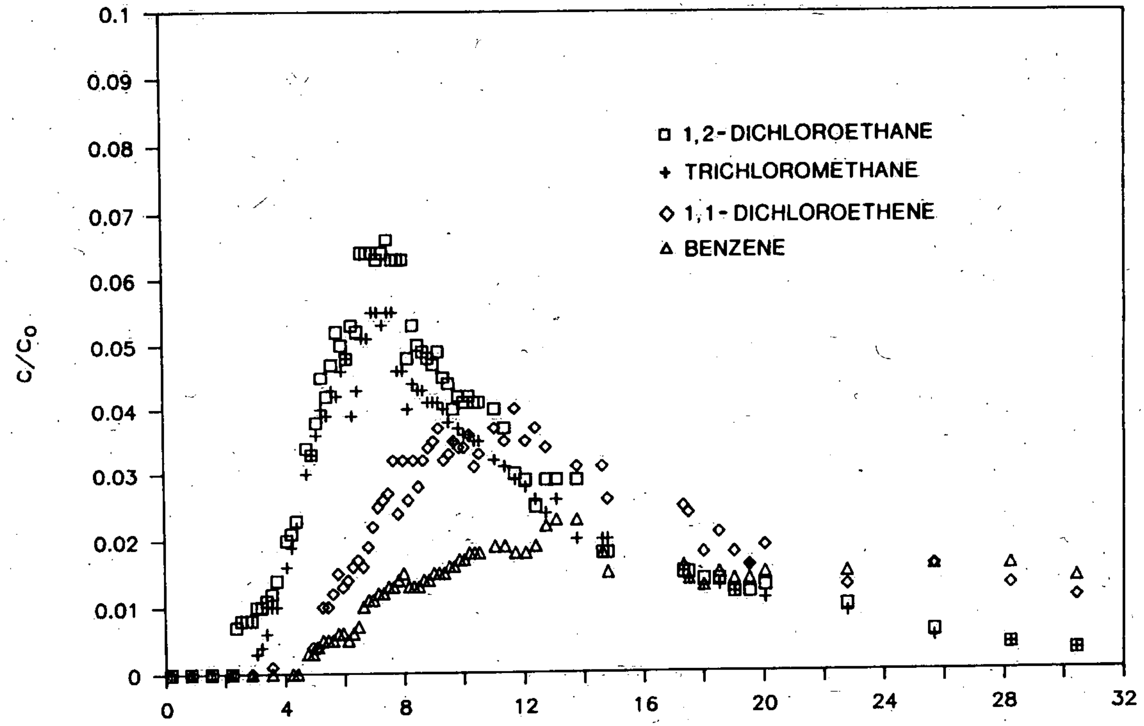


Figure 28. Breakthrough curves of 1,2-dichloroethane, chloroform or trichloromethane, 1,1-dichloroethene, and benzene for the same test as Figure 27.

A number of tests were run using the apparatus shown in Figure 26 with a sample of the Gloucester outwash aquifer taken from 15 m deep, 100 m east of the SWC. The column was packed to a bulk density of 1.76 g/cm^3 with a porosity of 0.32 and a pore volume of 205 cm^3. Analysis of 20 samples of the aquifer material indicated a mean organic carbon content of 0.07% and a standard deviation of 0.03%. This value is considerably lower than those reported for the Gloucester outwash aquifer materials by Jackson *et al.* (1985); these earlier values are now thought to be incorrect.

Results from one of the column tests (#3) are shown in Figures 27 and 28 and summarized in Table 12. Ground water flowed through the column with an average linear velocity of 45 cm/d. The length of the pulse of the organic tracers was 0.65 pore volumes. Estimates of R_f taken from the pore volume value when $C/C_0 = 0.5$ are low compared with the fitted results from the nonequilibrium model (CXTFIT, model 4) of Parker and van Genuchten (1984). Because of the failure of this model to reproduce the examples in Parker and van Genuchten, these latter values are considered suspect. A code to compute the centre of mass (CofM) of the BTC gave estimates midway between those two values. These CofM values are considered to be the most reliable on the basis of their similarity to the estimates of Patterson *et al.* (1985) for trichloromethane (TCM or chloroform) and 1,1-dichloroethene (11DCE), compounds not used by Patterson *et al.* to derive their predictive equation.

The method of Schwarzenbach and Westall (1981) gives poor estimates when compared to the field R_f values. This may be expected because the method was developed from correlations with compounds having log K_{ow} values from 2.6 to 4.7 and for aquifer materials with at least 0.1% organic carbon. Furthermore, it is evident from Figure 28 that the compounds elute in approximately the order of their increasing log K_{ow} value. However, coelution of 1,2-dichloroethane (12DCA) and trichloromethane occurs despite their log K_{ow} values being one-half log unit different. Finally, the LEA model (#2) of Parker and van Genuchten (1984) provides better estimates of R_f for the only slightly retarded aliphatic ethers with their symmetric BTCs (see Fig. 27) than the nonequilibrium model (#4) does for the strongly retarded VOCs, which display asymmetric BTCs as shown in Figure 28.

The data summarized in Table 12 suggests that laboratory column tests give reasonable estimates of the retardation factor for a organic solute, when compared to those estimated in the field.

CHAPTER 5

Discussion of Remedial Options

The alternatives for remedial action commonly used in the U.S.A. and Canada are listed in Table 13. These were considered by Graham Engineering Consultants and Water and Earth Sciences Ltd. (GEC 1985b) in their assessment of remedial measures at the Gloucester Landfill.

Of the management options, GEC considered the "no action" alternative unacceptable because the results of an endangerment assessment (Jackson *et al.* 1985) showed that the upper limit on lifetime cancer risk from drinking contaminated ground water from the site was much greater than 1 in 1000. GEC did recommend that alternative water supplies be developed for the community immediately east of the CPR tracks shown in Figure 6. In 1985, a pipeline was laid to supply these homes and industries with water from the City of Ottawa system.

A second management option recommended and implemented during 1988-89 was source removal. This involved the excavation of eight one-ton lugger boxes of bulk wastes consisting primarily of pesticide-contaminated soils and fifty-two 58-gallon (218-L) drums of liquid and solid chemical and biochemical wastes (MacLaren Engineers 1988).

Table 13. Corrective Action Alternatives (modified from OTA 1984)

1. Management options
    - limit/terminate aquifer use
    - develop alternative water supply
    - remove source of contamination
    - monitoring
    - no action
2. Containment
    - physical barriers, e.g., slurry walls, liners, clay, caps, etc.
    - hydraulic control via pumping and/or recharging
3. Withdrawal (and treatment)
    - pumping
    - excavation
    - gravity drainage
    - gas venting
4. In-situ restoration
    - biodegradation
    - steam injection
    - surfactant injection
    - enhanced solubilization

Containment alternatives considered included the installation of impermeable slurry walls or grout curtains around the zone of highest contamination. These techniques were rejected principally because the limestone bedrock underlying the plume is very permeable and is hydraulically connected to the overlying outwash aquifer (Jackson *et al.* 1985). In similar conditions at the Sylvester site in Nashua, New Hampshire, the remedial slurry wall and cap system was "much less effective than anticipated" (OTA 1985) due to leakage through the bedrock. Consequently complete isolation could not be assumed at the Gloucester Landfill.

Aquifer decontamination was selected as the most appropriate alternative. It involves a pump and treat scheme that will need to operate for an estimated five years and will involve four purge wells situated along the axis of the plume (GEC 1985b). Ground-water discharge will be treated by a bubbler system (to remove dissolved iron), air stripping and granular activated carbon adsorption. Treated water will be returned to the aquifer by five recharge wells. Tentative locations for the purge and recharge wells are shown in Figure 29. The four purge wells are planned to operate at a total rate of 120 gpm (655 m^3/d) based upon aquifer test results, which should ensure hydraulic containment of the plume (GEC 1985b). The cost of this operation is estimated to be about C$6 million (US$5 million) in current dollars.

Before a pump and treat operation is initiated, the presence of immiscible organic phases in the aquifer must be assessed. Such dense, nonaqueous phase liquids (DNAPLs) released by waste disposal operations or by spills tend to migrate to depth leaving residual droplets of liquid within the unsaturated and saturated zones. If DNAPLs are present in sufficient volume, pools may form on relatively impermeable strata within the aquifer or at its base (see Schwille 1984, 1988; Feenstra and Cherry 1988). Residual droplets ("ganglia") and pools are a long-term source of aquifer contamination. If they remain in an aquifer, decontamination may not be achieved by pump and

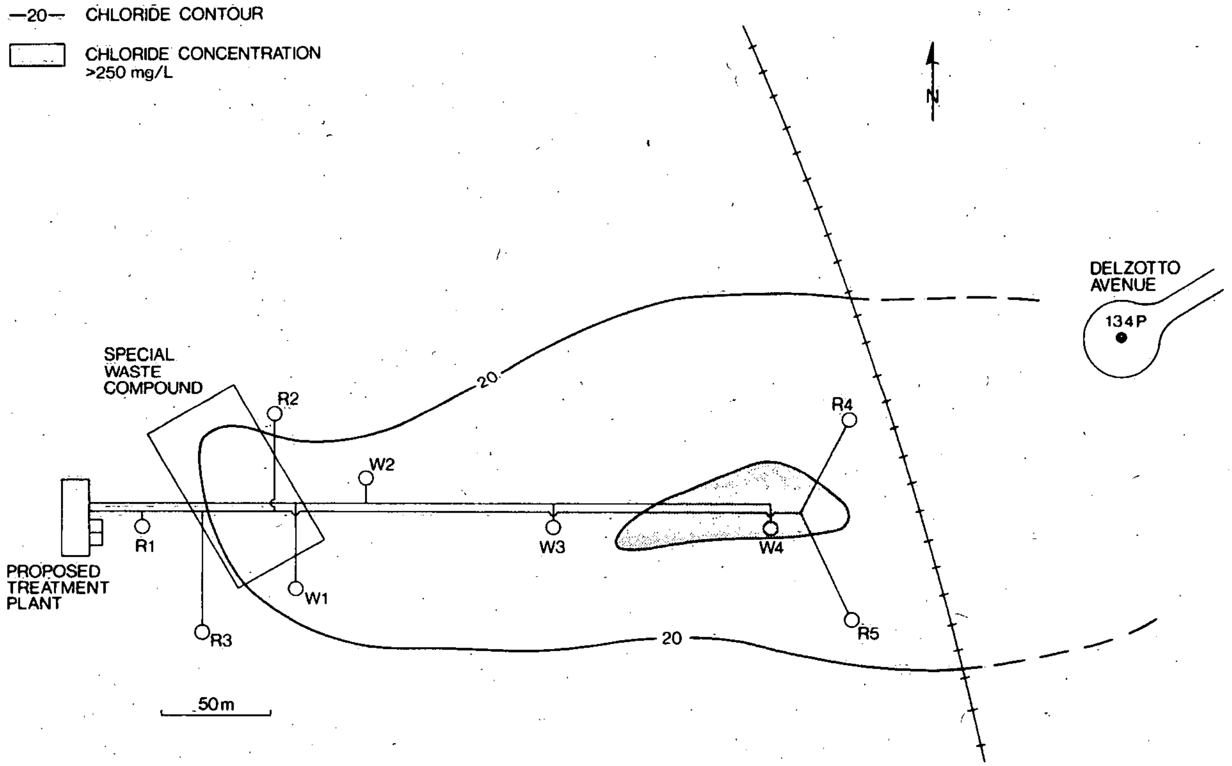


Figure 29. Geometry of the purge and recharge well network as recommended by GEC (1985b).

treat operations over the short term (Hunt *et al.* 1988a), or recontamination of a supposedly decontaminated aquifer may occur once the pump and treat operation is suspended.

As discussed in Chapter 3, the presence of DNAPLs beneath the SWC at Gloucester is uncertain. Concentrations of tetrachloroethene (PCE) in ground waters immediately beneath the SWC exceeded one percent of its aqueous solubility during 1982-83, as shown in Figure 23; this value is a rough indicator of the presence of DNAPLs in an aquifer (Hunt *et al.* 1988a). However, since that time, concentrations of most volatile organic contaminants have declined by a factor of about 100 (see Table 14 and Figs. 30-33), suggesting the absence of DNAPL pools. Furthermore, Figures 21 and 22 show no increase in contamination with depth as would be expected should a DNAPL pool have sunk through the aquifer. In addition, monitoring well 37P-3 in the underlying bedrock remains free of contamination despite being hydraulically connected to the outwash aquifer. Therefore, it seems entirely possible that DNAPLs in limited quantities did enter the outwash aquifer in 1982-83 or before and still exist as emulsified droplets or ganglia but not as pools.

Experimental pump and treat operations were conducted during 1984 and 1988 in the SWC with withdrawal and recharge wells completed in the outwash aquifer beneath the SWC. Results suggest that the residual contaminants can be removed from the aquifer, although not necessarily from the overlying aquitard, which may require excavation. Figures 30-33 show the substantial reduction in benzene, chloroform, 1,1,1-trichloroethane, and trichloroethene between 1984 and 1988.

To demonstrate the effectiveness of decontaminating the outwash aquifer by purge wells (i.e., the pump and treat option), the purge well test was conducted (Whiffin and Bahr 1985; Bahr 1989) at a site (54M) near the centre of the contaminant plume. As discussed in Chapter 4, this test involved the introduction of uncontaminated ground water into the contaminated aquifer via an injection well and the withdrawal of a similar volume of contaminated ground water from another well 5 m away. Operation of this system resulted in the purging of the contaminated aquifer between the two wells (see Fig. 24). A second test of this type was conducted in a shallow plume of organic contaminants in the unconfined aquifer (Unit E) with similar results (WESA 1988).

Analysis of the disappearance curves in Figure 24 indicates that there is a log-linear relationship (Fig. 34) between the number of pore volumes withdrawn from the aquifer to obtain 90% apparent decontamination of each of the organics and their respective K_{ow} values. This site-specific relationship can be extrapolated (i.e., beyond ten pore volumes) to obtain an estimate of the number of pore volumes required to decontaminate

Table 14. Volatile Organic Compounds Measured by GC/MS in μg/L in Ground Waters from Selected Multilevel Samplers within the Outwash Aquifer (see Fig. 6 for locations)

Compound	16M-16		37M-9	
	1982	1988	1982	1988
Dichloromethane	620	30	6 050	25
Chloroform	12 900	129	32 000	24
Bromoform	90	n.d.	n.d.	n.d.
Vinyl chloride	n.m.	n.d.	n.m.	40
1,1-Dichloroethene	20	n.d.	30	60
1,2-Dichloroethane	480	3.9	2 700	25
1,1,1-Trichloroethane	1 900	141	3 500	176
Trichloroethene	1 600	38	20	110
Tetrachloroethene	4 500	62	n.d.	n.d.
Benzene	580	6	17 000	646

n.d. = Not detectable.
n.m. = Not measured.

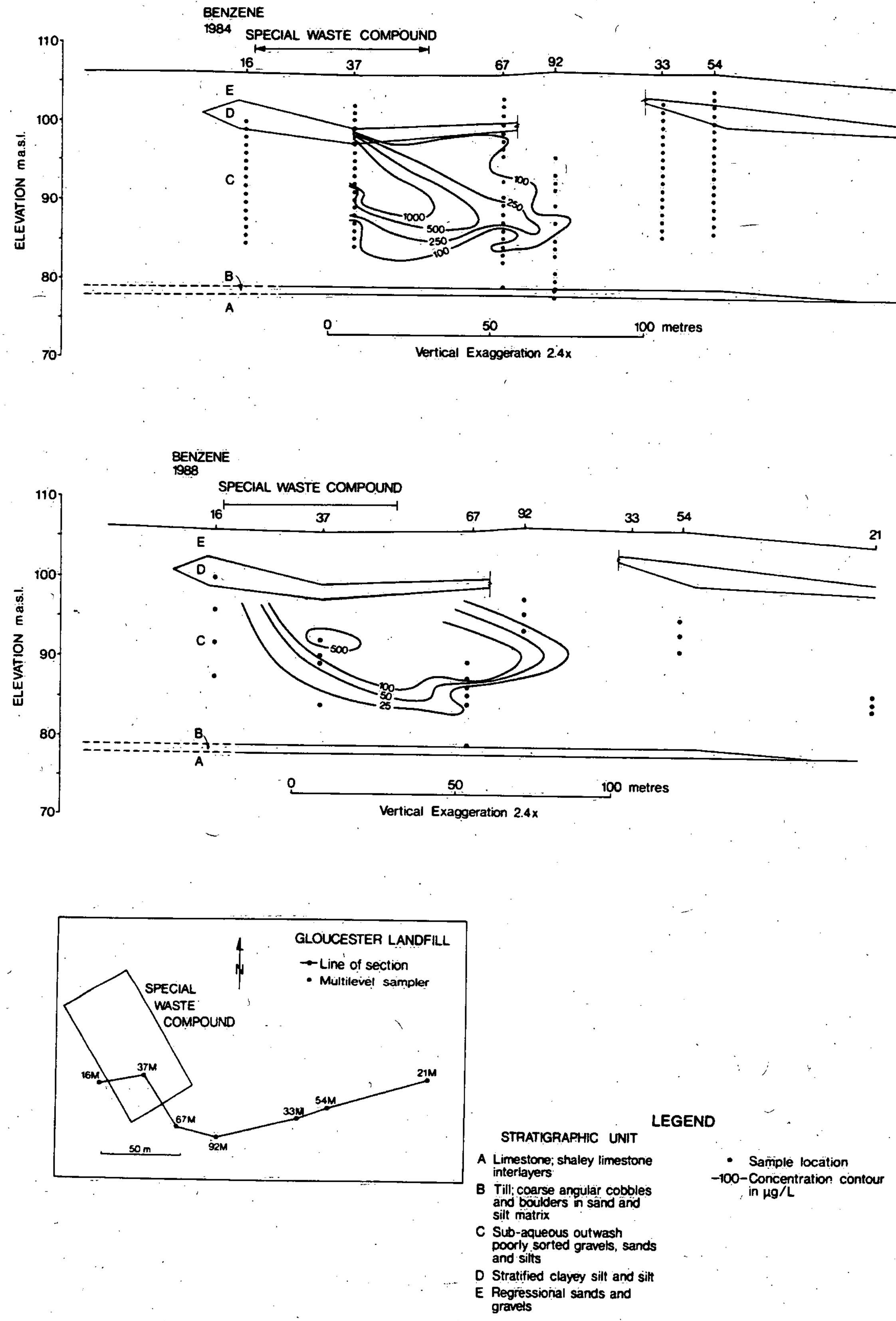


Figure 30. Cross-sectional profile of benzene contamination in 1984 and 1988.

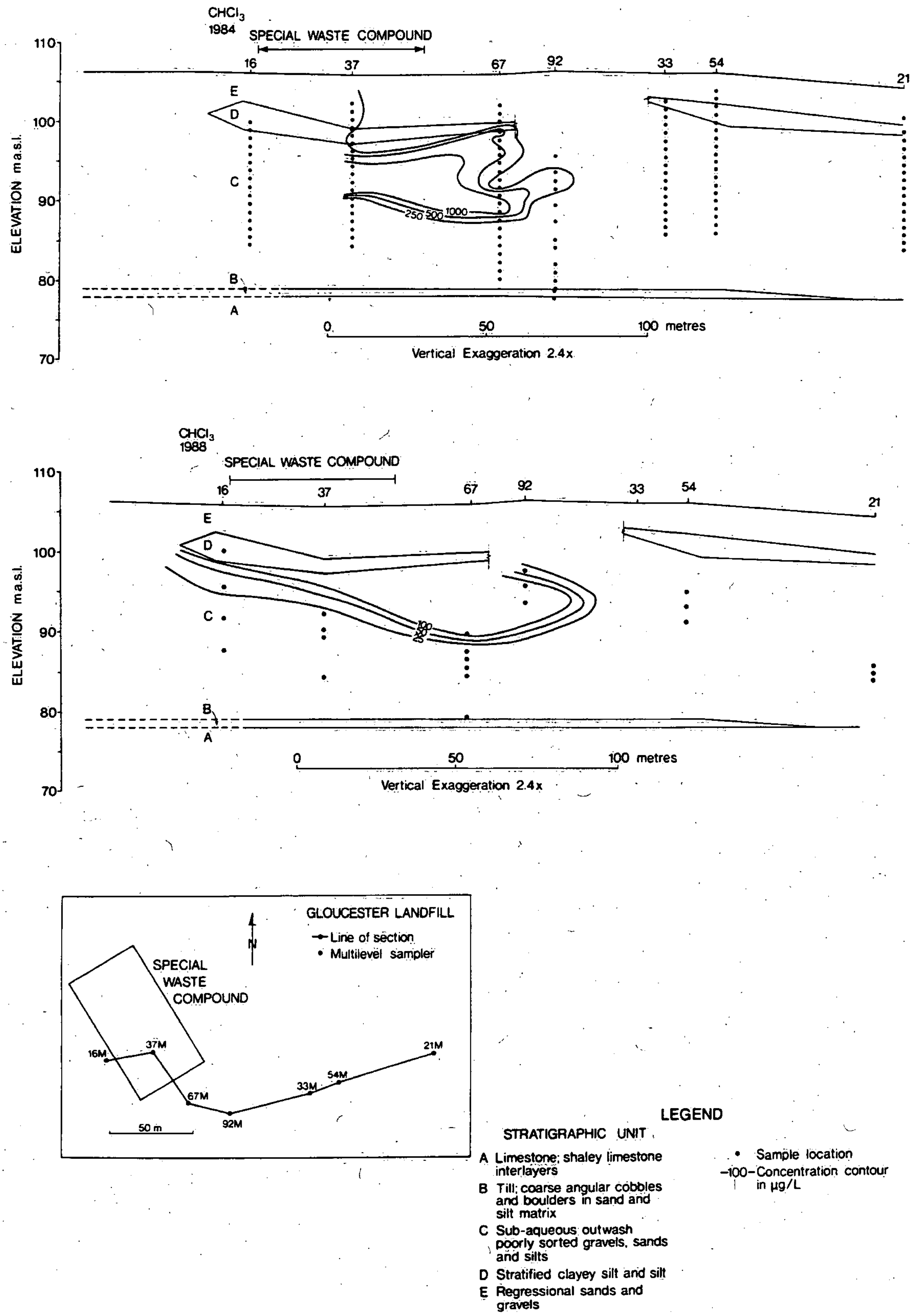


Figure 31. Cross-sectional profile of chloroform contamination in 1984 and 1988.

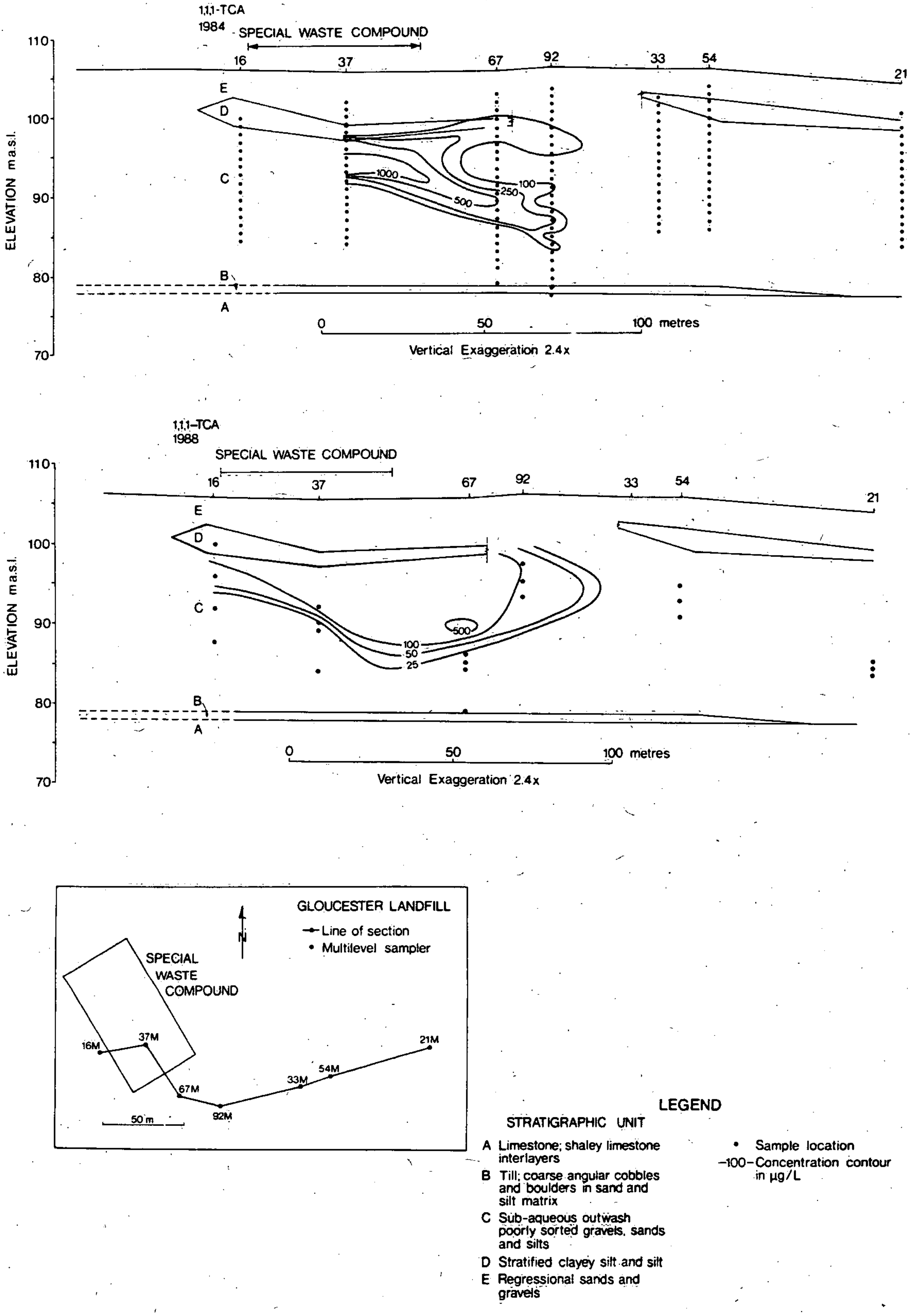


Figure 32. Cross-sectional profile of 1,1,1-trichloroethane contamination in 1984 and 1988.

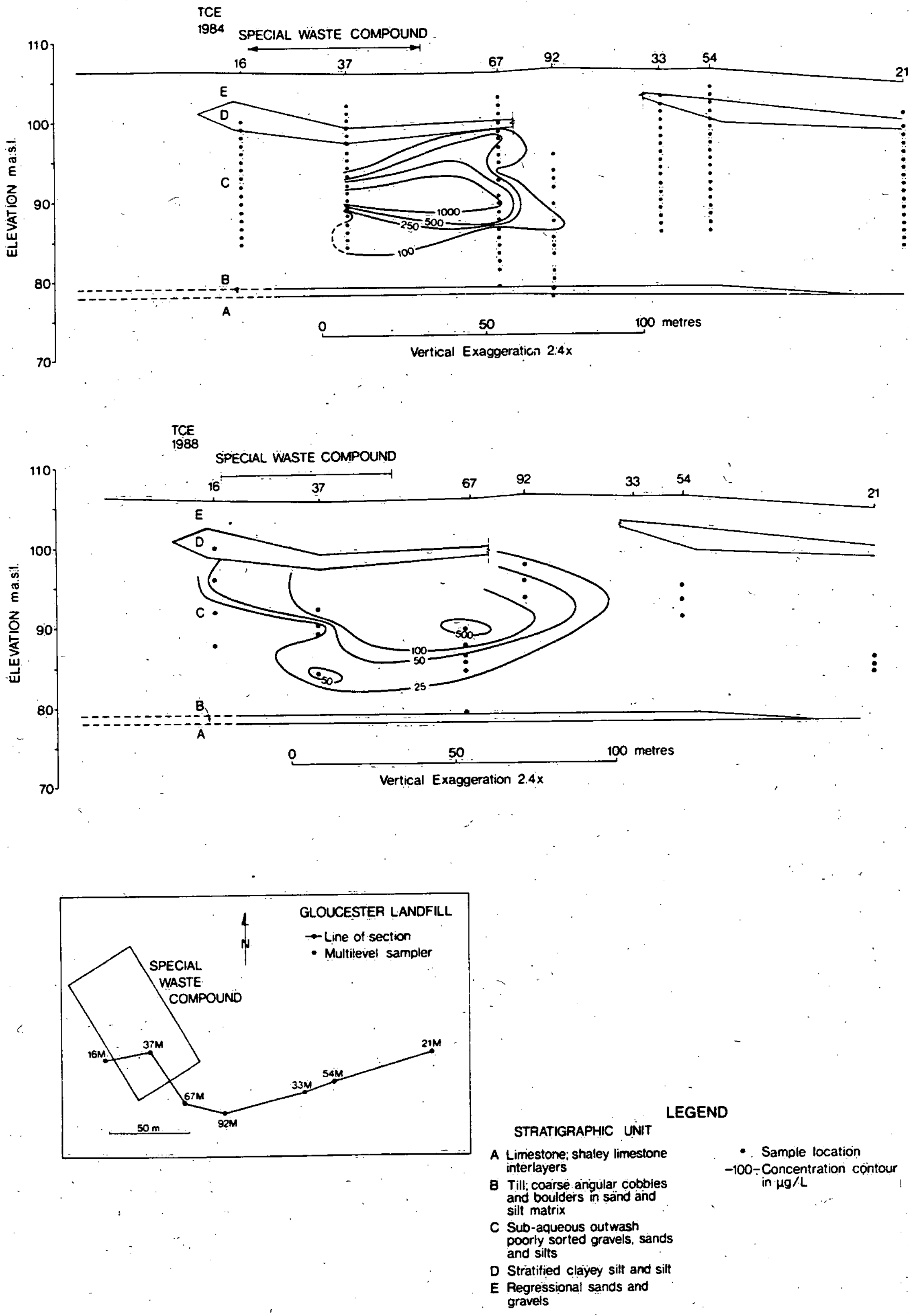


Figure 33. Cross-sectional profile of trichloroethene contamination in 1984 and 1988.

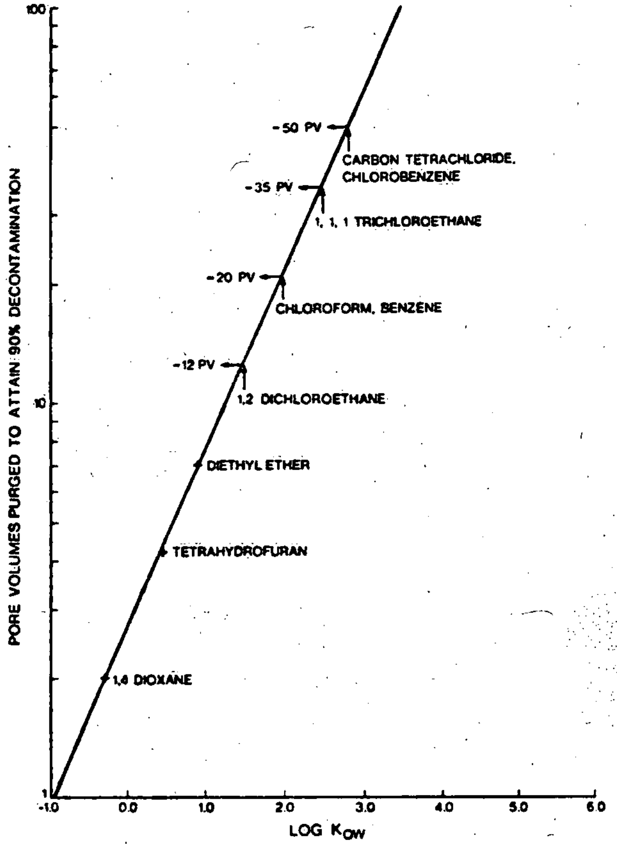


Figure 34. Estimation of the number of pore volumes that must be purged from the outwash aquifer to achieve 90% decontamination. Pore volume values for diethyl ether, tetrahydrofuran and 1,4-dioxane only are obtained from Fig. 24. Other values are extrapolated.

parts of the aquifer containing contaminants not found in the test area. Note that the values in Figure 34 are minimum estimates of the number of pore volumes required to decontaminate this particular aquifer under the hydraulic conditions of the test. Because local chemical equilibrium was not attained everywhere during the test, large volumes of relatively dilute ground water were pumped from the aquifer. Bahr (1989) has concluded that "pumping may be more efficient, in terms of volumes of clean water required for injection and volumes of contaminated water removed for treatment, if lower flow rates are used." By so doing, more time is available to allow the relatively slow desorption of the organic chemicals.

Because of the high cost of pump and treat operations, there is much interest in developing lower cost alternatives. The most promising of these is in situ biological remediation. This technique, generally known as biorestoration, involves the prior removal of DNAPL pools and the subsequent injection of oxygen and nutrients into a plume of dissolved aromatic hydrocarbons to stimulate the native microbial populations to metabolize or cometabolize the organic contaminants (Wilson *et al.* 1986; Lee *et al.* 1988b). Because in-situ biorestoration of halogenated hydrocarbons is still under development (Semprini *et al.* 1988), it was not considered as a remedial option for the Gloucester aquifer. However, towards the end of the five years of purging the contaminated aquifer, in situ biorestoration may become available to clean up the residual contamination that will undoubtedly remain. The biological techniques are potentially cost-effective methods for reducing residual contamination to compliance levels, which, in this case, are similar to EPA's maximum contaminant levels.

Other in-situ techniques, such as steam injection (Hunt *et al.* 1988a, 1988b), the use of surfactants (Vigon and Rubin 1989), and enhanced solubilization (Zenon 1986), all show promise but as yet are unproven.

CONCLUSIONS

An exhaustive assessment has been conducted of the organically polluted outwash aquifer beneath the Gloucester Landfill, and potentially applicable alternative strategies for remediation have been evaluated (Jackson et al. 1989; Jackson and Patterson 1989). Our conclusions are that

1. the installation of a slurry wall system to contain the organic pollutants is unsuitable because of the very permeable nature of the bedrock;
2. pools of DNAPL chemicals are unlikely to be present within the outwash aquifer although emulsified DNAPL chemicals probably still exist as ganglia;
3. the aquitard immediately beneath the SWC may contain substantial quantities of DNAPLs and should be investigated with regard to its possible excavation;
4. a pump and treat operation should be successful in removing most of the dissolved and sorbed organic chemicals in the outwash aquifer provided that the purge and recharge wells are placed in the optimal locations; and
5. in-situ biorestoration may be required to complete the decontamination of the aquifer following the pump and treat operations.

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Appendix

Complete 1984 and 1988 Data for Volatile Organic Chemicals in the Outwash Aquifer, Gloucester Landfill

(by D. Chaput and R.J. Patterson, NHRI)

Table A-1. 1984 Volatile Organic Analysis

Sampling point	DEE µg/L	11-DCE µg/L	MECL µg/L	T12-DE µg/L	11-DCA µg/L	ACET µg/L	CHCL3 µg/L	THF µg/L	111-TCA µg/L	12-DCA µg/L	BENZ µg/L	CCL4 µg/L	TCE µg/L	14DNE µg/L	TOL µg/L	PCE µg/L	CLBEN µg/L	ETBEN µg/L	CHBR3 µg/L
33M1	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M2	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M3	0.6	0.0	0.0	0.0	0.0	0.0	0.0	23.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M6	156.3	0.0	0.0	0.0	0.0	0.0	0.0	15.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M8	249.4	0.0	0.0	0.0	0.0	0.0	0.0	47.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M9	26.7	0.0	0.0	0.0	0.0	0.0	0.0	46.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M10	17.1	0.0	0.0	0.0	0.0	0.0	0.0	24.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M11	9.3	0.0	0.0	0.0	0.0	0.0	0.0	4.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M12	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M13	26.4	0.0	0.0	0.0	0.0	0.0	0.0	12.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M14	3.2	0.0	0.0	0.0	0.0	0.0	0.0	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33M15	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
33M16	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
34M2	368.7	0.0	0.0	0.0	0.0	0.0	0.0	352.8	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34M3	878.3	0.0	0.0	0.0	0.0	0.0	0.0	669.0	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34M4	754.5	0.0	0.0	0.0	0.0	0.0	0.0	288.7	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34M5	951.4	0.0	0.0	0.0	0.0	0.0	0.0	414.5	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34M6	3.2	0.0	0.0	0.0	0.0	0.0	0.0	277.3	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34M7	2.1	0.0	0.0	0.0	0.0	0.0	0.0	3.9	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34M8	1.8	0.0	0.0	0.0	0.0	0.0	0.0	4.2	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34M9	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34M11	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	P	0.4	0.0	0.0	0.0	0.0
34M12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	P	0.5	0.0	0.0	0.0	0.0
34M18	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0
37M1	0.0	4.6	3.4	0.0	5.5	0.0	31.7	0.0	7.6	1.6	96.5	4.8	22.2	0.0	1.2	1.2	0.1	0.6	0.0
37M2	2.0	5.7	3.0	0.0	4.8	0.0	27.0	0.0	5.0	1.1	91.5	3.5	17.1	0.0	1.0	0.9	0.0	0.1	0.0
37M3	4.4	3.5	1.6	0.0	3.3	0.0	16.5	7.9	4.4	1.0	60.2	2.4	13.2	0.0	1.0	1.0	0.0	0.2	0.0
37M4	111.2	<5.00	0.6	0.0	15.1	0.0	12.0	19.2	2.5	18.6	43.1	1.9	10.4	0.0	0.7	0.8	0.0	0.1	0.0
37M5	292.6	<5.00	0.5	0.0	38.6	0.0	14.1	130.5	2.2	36.0	320.2	2.2	11.0	0.0	1.2	0.6	0.0	0.1	0.0
37M6	191.0	<5.00	1.9	0.0	58.0	0.0	18.3	48.2	1.9	35.3	536.5	2.0	11.2	0.0	1.7	0.8	0.0	0.6	0.0
37M7	406.5	8.5	3.6	0.0	106.6	0.0	14.4	235.2	0.9	96.9	988.8	2.3	9.4	0.0	4.6	0.7	0.0	0.7	0.0
37M8	850.7	27.5	440.0	0.0	662.0	0.0	7265.7	865.4	10.5	555.8	500.0	10.2	14.9	0.0	2.6	1.9	0.0	0.8	0.0
37M9	979.5	144.0	860.0	0.0	1416.3	0.0	9632.9	874.5	158.8	922.5	3394.8	14.9	36.7	0.0	4.7	3.6	0.0	1.6	0.0
37M10	558.5	703.0	754.0	0.0	1768.3	0.0	9645.9	485.6	1182.0	847.9	3182.8	6.9	220.2	0.0	8.2	3.3	0.0	1.1	0.0
37M11	373.5	1093.0	578.0	0.0	1591.4	0.0	4556.5	164.3	2234.8	475.3	2312.3	15.2	697.2	0.0	19.6	0.0	0.0	1.4	0.0
37M12	164.2	747.0	150.0	0.0	1571.0	0.0	723.5	0.0	1096.0	201.0	1690.4	8.6	1525.1	0.0	35.5	0.0	0.0	0.0	0.0
37M13	138.8	785.0	100.0	0.0	1257.1	0.0	335.6	0.0	802.7	134.1	1473.8	8.1	1616.7	0.0	585.9	0.0	0.0	0.0	0.0
37M14	124.9	530.0	T	0.0	673.1	0.0	97.0	0.0	652.6	99.8	1562.1	5.0	371.6	0.0	10.7	0.0	0.0	0.0	0.0
37M15	68.3	80.0	5.0	8.7	89.1	0.0	144.9	0.0	15.2	59.1	514.2	0.0	17.4	0.0	4.0	0.0	0.0	0.0	0.0
37M16	19.1	22.4	14.5	0.0	10.0	0.0	423.0	0.0	34.7	18.8	48.9	123.4	169.7	0.0	1.4	14.2	2.8	0.0	143.3
37M17	18.4	12.2	12.7	0.0	8.0	0.0	358.3	0.0	32.9	18.8	35.3	128.7	113.0	0.0	3.2	13.1	3.3	0.0	95.0
37M18	22.9	11.3	17.3	0.0	6.6	0.0	378.3	15.5	33.0	21.5	38.7	170.0	130.6	0.0	1.6	19.7	4.5	0.0	120.2
37M19	26.3	9.1	19.6	0.0	5.4	0.0	359.2	13.7	26.2	19.3	30.5	179.4	134.2	0.0	0.0	16.1	3.2	0.0	140.0
38M2	45.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.8	2.6	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
38M3	23.8	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	7.9	2.3	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
38M4	162.2	0.0	0.0	0.0	0.4	0.0	0.0	80.0	0.0	16.4	2.6	0.0	0.3	P	0.0	0.0	0.0	0.0	0.0
38M6	308.4	0.0	0.0	0.0	0.4	0.0	0.0	22.1	0.0	11.1	2.5	0.0	0.4	P	0.0	0.0	0.0	0.0	0.0
38M8	80.1	0.0	0.0	0.0	0.3	0.0	0.0	2.2	0.0	4.8	0.2	0.0	0.2	P	0.0	0.0	0.0	0.0	0.0

Table A-1. Continued.

Sampling point	DEE µg/L	11-DCE µg/L	MECL µg/L	T12-DE µg/L	11-DCA µg/L	ACET µg/L	CHCL3 µg/L	THF µg/L	111-TCA µg/L	12-DCA µg/L	BENZ µg/L	CCL4 µg/L	TCE µg/L	14DNE µg/L	TOL µg/L	PCE µg/L	CLBEN µg/L	ETBEN µg/L	CHBR3 µg/L
38M9	160.9	0.0	0.0	0.0	0.2	0.0	0.0	2.8	0.0	1.7	1.2	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
38M11	112.8	0.0	0.0	0.0	0.5	0.0	0.0	129.5	0.0	1.7	12.9	0.0	3.4	P	0.0	0.0	0.0	0.0	0.0
38M13	455.5	0.0	0.0	0.0	2.3	0.0	4.5	215.5	0.6	3.7	28.7	0.0	18.8	P	0.3	0.0	0.0	0.0	0.0
38M15	568.8	0.0	0.0	0.0	23.4	0.0	0.0	445.4	0.0	26.6	122.1	0.0	6.7	P	0.0	0.0	0.0	0.0	0.0
38M17	501.9	0.0	0.0	0.0	13.2	0.0	0.8	180.7	0.0	9.4	62.0	0.0	3.0	P	0.0	0.0	0.0	0.0	0.0
38M18	21.1	0.0	0.0	0.0	0.5	0.0	0.5	3.3	0.0	2.5	4.4	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0
56M4	71.6	0.0	1.0	0.0	7.6	0.0	1.1	268.7	0.0	19.5	110.1	0.0	12.9	0.0	2.5	0.6	6.4	41.2	0.0
56M7	1.9	0.0	0.3	0.0	6.2	0.0	1.0	23.7	0.0	0.0	6.1	0.0	3.0	0.0	12.8	0.0	0.0	6.9	0.0
67M1	0.0	2.1	0.0	0.0	0.8		5.2		5.1	0.0	10.7	0.0	38.0	0.0	0.0	0.0	0.0	0.0	0.0
67M3	0.0	2.1	0.0	0.0	2.0		1.8		1.5	2.5	38.7	0.0	24.5	0.0	0.0	0.0	0.0	0.0	0.0
67M4	0.0	2.6	0.0	0.0	3.4		11.8		17.7	3.3	45.1	0.0	62.3	0.0	0.0	0.0	0.0	0.0	0.0
67M5	0.0	1.2	0.0	0.0	2.2		1.9		1.2	1.1	31.0	0.0	31.4	0.0	0.0	0.0	0.0	0.0	0.0
67M6	0.0	1.4	0.0	0.0	14.4		4.1		11.2	8.5	154.0	0.0	41.9	0.0	0.0	0.0	0.0	0.0	0.0
67M7	0.0	1.4	0.0	0.0	9.8		1.1		12.1	4.5	90.4	0.0	30.4	0.0	0.0	0.0	0.0	0.0	0.0
67M8	0.0	0.0	0.0	0.0	1.5		0.0		1.0	0.7	27.3	0.0	23.5	0.0	0.0	0.0	0.0	0.0	0.0
67M9	79.0	0.0	0.0	0.0	155.0		15.2		263.0	50.3	418.0	0.0	854.0	0.0	0.0	0.0	0.0	0.0	0.0
67M10	108.0	P			234.0		131.0		300.0	105.0	372.0	0.0	694.0	0.0	0.0	0.0	0.0	0.0	0.0
67M11	100.0	P	?	0.0	112.0		2885.0		645.0	>200.0	289.0	0.0	1561.0	0.0	0.0	0.0	0.0	0.0	0.0
67M12	62.0		0.0	0.0	69.0		1754.0		400.0	200.0	228.0	0.0	742.0	0.0	0.0	0.0	0.0	0.0	0.0
67M14	35.0		0.0	0.0	63.0		171.0		69.0	30.0	192.0	0.0	253.0	0.0	0.0	0.0	0.0	0.0	0.0
67M15	14.0		0.0	0.0	14.0		168.0		16.0	11.0	96.0	0.0	62.0	0.0	0.0	0.0	0.0	0.0	0.0
67M16	11.0		0.0	0.0	25.0		324.0		65.0	14.0	79.0	0.0	162.0	0.0	0.0	0.0	0.0	0.0	0.0
67M17	5.7		0.0	0.0	22.0		470.0		86.0	22.0	36.0	0.0	149.0	0.0	0.0	0.0	0.0	0.0	0.0
67M18	10.0		0.0	0.0	23.0		5.8		89.0	26.0	52.0	0.0	181.0	0.0	0.0	0.0	0.0	0.0	0.0
67M19	17.0		0.0	0.0	97.0		803.0		183.0	44.0	123.0	0.0	272.0	0.0	0.0	0.0	0.0	0.0	0.0
67M20	7.0		0.0	0.0	23.0		452.0		190.0	18.0	33.0	15.0	170.0	0.0	0.0	0.0	0.0	0.0	0.0
67M21	0.0		0.0	0.0	8.7		894.0		202.0	18.0	T	29.0	36.0	0.0	0.0	0.0	0.0	0.0	0.0
67M22	0.0		0.0	0.0	2.7		101.0		59.0	<2.0	0.0	7.8	18.0	0.0	0.0	0.0	0.0	0.0	0.0
67M23	0.0		0.0	0.0	1.8		22.0		21.0	<2.0	T	T	11.0	0.0	0.0	0.0	0.0	0.0	0.0
67M24	9.0		0.0	0.0	9.0		37.0		38.0	9.0	22.0	0.0	72.0	0.0	0.0	0.0	0.0	0.0	0.0
92M1	0.7	0.6	0.0	0.0	0.6	0.0	1.0	0.0	1.6	0.3	1.7	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0
92M2	1.7	1.8	0.0	0.0	5.2	0.0	1.4	0.0	3.7	1.1	9.4	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0
92M3	4.1	2.1	0.0	0.0	11.1	0.0	2.4	0.0	4.7	2.2	11.1	0.0	2.2	0.0	0.0	0.0	0.0	0.0	0.0
92M5	13.1	35.5	0.0	0.0	32.3	0.0	4.3	0.0	33.8	10.0	37.3	0.0	9.1	0.0	0.0	0.0	0.0	0.0	0.0
92M6	0.0	41.5	0.0	1.4	35.5	0.0	36.0	5.9	106.4	9.9	74.6	0.0	46.5	O.O	0.0	0.0	0.0	0.0	0.0
92M7	15.6	31.2	0.0	0.0	25.6	0.0	40.7	30.9	77.3	8.6	71.1	0.0	43.6	0.0	0.0	0.0	0.0	0.0	0.0
92M9	48.3	245.9	0.0	0.0	176.4	0.0	105.9	55.0	253.3	53.8	211.1	0.0	154.8	0.0	0.0	0.0	0.0	0.0	0.0
92M11	16.0	15.0	0.0	0.0	15.5	0.0	42.3	0.0	35.4	10.3	38.3	0.0	22.5	0.0	0.0	0.0	0.0	0.0	0.0
92M13	11.0	73.9	0.0	0.0	16.0	0.0	472.2	0.0	144.1	27.4	74.5	0.0	52.8	0.0	0.0	0.0	0.0	0.0	0.0
92M14	15.6	34.4	0.0	1.8	18.9	0.0	272.7	0.0	75.5	19.9	44.7	0.0	39.0	0.0	0.0	0.0	0.0	0.0	0.0
92M15	9.3	38.9	0.0	0.9	15.0	0.0	310.1	10.5	87.8	18.2	42.5	0.0	46.1	0.0	0.0	0.0	0.0	0.0	0.0
92M17	0.0	66.4	0.0	0.0	28.2	0.0	179.4	0.0	190.8	0.0	26.3	0.0	30.7	0.0	0.0	0.0	0.0	0.0	0.0
93M15	713.9	130.9	5.5	0.0	50.3	0.0	48.1	68.5	108.2	31.2	164.4	0.0	54.0	0.0	1.6	0.0	0.8	0.0	0.0
93M17	484.7	82.2	5.5	0.0	130.1	0.0	147.8	90.7	287.0	64.7	387.0	0.0	77.0	0.0	0.0	0.0	0.0	0.0	0.0
93M19	26.6	5.9	5.2	0.0	26.1	0.0	26.7	18.1	22.3	22.5	164.5	0.0	5.2	0.0	0.0	0.0	0.0	0.0	0.0
93M20	11.9	6.5	1.8	0.0	8.7	0.0	9.2	0.0	14.3	6.7	59.7	0.0	6.5	0.0	0.0	0.0	0.0	0.0	0.0
93M21	8.2	7.3	1.0	0.0	7.0	0.0	6.0	0.0	12.3	5.7	43.7	0.0	3.4	0.0	0.0	0.0	0.0	0.0	0.0

Table A-1. Continued.

Sampling point	DEE µg/L	11-DCE µg/L	MECL µg/L	T12-DE µg/L	11-DCA µg/L	ACET µg/L	CHCL3 µg/L	THF µg/L	111-TCA µg/L	12-DCA µg/L	BENZ µg/L	CCL4 µg/L	TCE µg/L	14DNE µg/L	TOL µg/L	PCE µg/L	CLBEN µg/L	ETBEN µg/L	CHBR3 µg/L
93M2	3.6	12.0	0.5	0.0	12.9	0.0	1.9	0.0	25.7	0.0	4.2	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
94M2	252.9	0.0	0.0	0.0	0.0	0.0	0.0	49.9	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94M5	1000.0	0.0	0.0	0.0	0.0	0.0	0.5	96.8	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94M6	641.6	0.0	0.0	0.0	0.0	0.0	0.3	229.4	0.2	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94M9	1254.4	0.0	0.0	0.0	0.0	0.0	0.0	573.0	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94M10	974.4	0.0	0.0	0.0	0.0	0.0	0.0	819.7	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94M11	1022.8	0.0	0.0	0.0	0.0	0.0	0.0	799.0	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94M12	1152.5	0.0	0.0	0.0	0.0	0.0	1.0	1059.2	0.0	0.3	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94M18	1113.4	0.0	0.0	0.0	14.0	0.0	0.0	210.5	2.0	9.7	159.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
94M21	747.5	0.0	0.0	0.0	0.9	0.0	0.0	171.0	0.0	0.8	8.3	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94M22	89.4	0.0	0.0	0.0	0.3	0.0	0.0	5.6	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
94M23	1.3	0.0	0.0	0.0	0.8	0.0	0.4	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
94M24	10.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.9	0.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95M1	1.1	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.5	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95M2	8.7	1.1	0.0	0.0	0.5	0.0	0.0	0.0	0.0	1.0	0.8	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
95M3	35.1	0.4	0.0	0.0	0.4	0.0	2.5	4.9	0.8	0.0	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
95M5	254.8	36.7	0.0	0.0	22.4	0.0	4.3	104.6	4.0	3.3	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95M6	232.2	2.5	1.5	0.0	36.9	0.0	41.8	133.2	2.1	6.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95M7	201.1	3.4	3.7	0.0	58.9	0.0	108.9	90.1	4.6	6.9	4.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95M9	211.0	66.7	11.1	0.0	99.5	0.0	177.4	114.1	149.5	18.5	34.3	0.0	3.2	0.0	0.0	0.0	0.0	0.0	0.0
95M11	91.3	56.8	11.1	0.0	53.0	0.0	164.6	10.0	152.0	15.9	41.4	0.0	3.9	0.0	0.0	0.0	0.0	0.0	0.0
95M13	17.8	44.9	44.4	0.0	33.9	0.0	158.0	0.0	125.5	13.8	43.1	0.0	4.2	0.0	0.0	0.0	0.0	0.0	0.0
95M14	22.2	74.7	55.6	0.0	38.9	0.0	589.4	0.0	186.7	27.4	68.5	0.0	7.6	0.0	0.0	0.0	0.0	0.0	0.0
95M15	36.1	100.0	100.0	0.0	39.7	0.0	1107.9	46.3	356.6	100.0	116.0	0.0	20.9	0.0	0.0	0.0	0.0	0.0	0.0
95M17	22.2	124.2	41.1	0.0	36.1	0.0	669.1	0.0	270.7	77.8	68.7	0.0	4.9	0.0	0.0	0.0	0.0	0.0	0.0
95M18	26.4	57.2	23.4	0.0	62.7	0.0	381.1	0.0	176.7	18.6	45.0	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0
95M19	22.2	123.8	88.9	0.0	38.4	0.0	614.8	0.0	308.2	32.2	66.0	0.0	8.7	0.0	0.0	0.0	0.0	0.0	0.0
95M20	18.4	38.7	14.7	0.0	73.3	0.0	266.5	0.0	107.8	14.2	34.1	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0
95M21	12.6	38.2	12.2	0.0	97.8	0.0	117.5	0.0	68.6	12.3	31.4	0.0	2.3	0.0	0.0	0.0	0.0	0.0	0.0
95M22	38.2	3.3	5.5	0.0	76.2	0.0	22.8	0.0	16.5	6.7	12.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95M23	25.9	2.2	3.2	0.0	36.4	0.0	16.6	19.8	11.6	4.3	7.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95M24	38.5	2.2	1.8	0.0	15.9	0.0	15.0	0.0	8.9	2.3	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Table A-2. 1988 Volatile Organic Analysis

(by S. Lesage, M.W. Priddle, and P. Riemann, NWRI)

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SAMPLE NUMBER	88GL124P	88GL126P	88GL127P	88GL128P	88GL12M14	88GL14BL2	88GL14M02
CONCENTRATION	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							
VINYL CHLORIDE	<1						
1,1-DICHLOROETHENE				<2			
DICHLOROMETHANE							
DIETHYL ETHER	643	331	122				
TETRAHYDROFURAN	70	230	180				30
TRANS-1,2-DICHLOROETHENE				<1			
1,1-DICHLOROETHANE	2.6			0.9			<3
CIS-1,2-DICHLOROETHENE				<1			
CHLOROFORM				<2			
1,1,1-TRICHLOROETHANE				<3			
CARBON TETRACHLORIDE				<4			
BENZENE	2.8					15	5.9
1,2-DICHLOROETHANE	10.3						
TRICHLOROETHYLENE							
1,2-DICHLOROPROPANE							
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE						16	<0.5
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE							
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE							
ETHYL BENZENE						6.2	
m+p-XYLENE							
o-XYLENE						2.7	
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE							
TRIMETHYL BENZENE							
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	93	87	88	72	94	93	91

* SURROGATE was spiked at 100 µg/L.

Table A-2. Continued.

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SAMPLE NUMBER	88GL14M03	88GL16M04	88GL16M08	88GL16M12	88GL16M16	88GL16MBL	88GL20M01
CONCENTRATION	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							
VINYL CHLORIDE							
1,1-DICHLOROETHENE				40		4.4	
DICHLOROMETHANE					30		
DIETHYL ETHER		<2	<3	<6	10		<3
TETRAHYDROFURAN	60				15		
TRANS-1,2-DICHLOROETHENE							
1,1-DICHLOROETHANE				6.6	2.9	5.5	
CIS-1,2-DICHLOROETHENE							
CHLOROFORM		4.2		27	129	4.9	<2
1,1,1-TRICHLOROETHANE				76	141	5.1	<3
CARBON TETRACHLORIDE							<2
BENZENE				12	6	6	
1,2-DICHLOROETHANE				6.6	3.9		
TRICHLOROETHYLENE				60	38		
1,2-DICHLOROPROPANE							
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE	<0.8					19	
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE					62		
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE							
ETHYL BENZENE						1.6	
m+p-XYLENE						10	
o-XYLENE						3	
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE					2.1		
TRIMETHYL BENZENE						1.6	
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	90	95	99	96	83	95	93

* SURROGATE was spiked at 100 µg/L.

Table A-2. Continued.

NATIONAL WATER RESEARCH INSTITUTE
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SAMPLE NUMBER	88GL20M10	88GL21M01	88GL21M02	88GL21M03	88GL21MBL	88GL37BL	88GL37M01
CONCENTRATION	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							
VINYL CHLORIDE	2.0						
1,1-DICHLOROETHENE						4	10
DICHLOROMETHANE							
DIETHYL ETHER	70	36	658	616	22		
TETRAHYDROFURAN			120	100			
TRANS-1,2-DICHLOROETHENE							<5
1,1-DICHLOROETHANE						32	4.7
CIS-1,2-DICHLOROETHENE							<5
CHLOROFORM							6.6
1,1,1-TRICHLOROETHANE						5.6	15
CARBON TETRACHLORIDE							7.2
BENZENE	3.0	<1	5	<1	5.7	54	13
1,2-DICHLOROETHANE						8.2	
TRICHLOROETHYLENE						5.4	54
1,2-DICHLOROPROPANE							
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE	1.7	<0.5	<0.5		17	13	0.8
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE							3.5
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE							1.1
ETHYL BENZENE					1.3	0.3	
m+p-XYLENE					9	10.3	
o-XYLENE					2.8	3.3	
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE						1.5	
2-ETHYLTOLUENE						0.3	
TRIMETHYL BENZENE					1.4	1.6	
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	92	93	95	92	97	80	85

* SURROGATE was spiked at 100 µg/L.

Table A-2. Continued.

NATIONAL WATER RESEARCH INSTITUTE
GROUNDWATER CHEMISTRY LABORATORY
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DATE: 89/07/12

SAMPLE NUMBER	88GL37M06	88GL37M07	88GL37M09	88GL47M01	88GL47M02	88GL47M04	88GL54BL
CONCENTRATION	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							
VINYL CHLORIDE			≃ 40	<1			
1,1-DICHLOROETHENE	35	3.8	60	<2			
DICHLOROMETHANE			25				
DIETHYL ETHER	41	172	254				22
TETRAHYDROFURAN							
TRANS-1,2-DICHLOROETHENE				<2			
1,1-DICHLOROETHANE	12	25	614	2.6	<3		
CIS-1,2-DICHLOROETHENE			7.5	<2			
CHLOROFORM	2.1		24			<2	
1,1,1-TRICHLOROETHANE			176	<4			
CARBON TETRACHLORIDE							
BENZENE	41	118	646		10.5	2.6	5.7
1,2-DICHLOROETHANE	5.2	14	25				
TRICHLOROETHYLENE	24	7.4	110				
1,2-DICHLOROPROPANE			11				
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE			2.5		<0.5	<0.3	18
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE	3.0	2.0					
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE							
ETHYL BENZENE							2.2
m+p-XYLENE							9.4
o-XYLENE							2.7
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE							
TRIMETHYL BENZENE							1.5
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	37	86	77	120	77	128	96

* SURROGATE was spiked at 100 µg/L.

Table A-2. Continued.

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GROUNDWATER CHEMISTRY LABORATORY
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DATE: 89/07/12

SAMPLE NUMBER	88GL54M06	88GL54M08	88GL54M10	88GL67M01	88GL67M06	88GL67M07	88GL67M08
CONCENTRATION	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							
VINYL CHLORIDE							
1,1-DICHLOROETHENE				1.2	0.9		
DICHLOROMETHANE							
DIETHYL ETHER	250	269	258				
TETRAHYDROFURAN	30	45	20				
TRANS-1,2-DICHLOROETHENE							
1,1-DICHLOROETHANE				2.1	3.1		
CIS-1,2-DICHLOROETHENE							
CHLOROFORM				2.8	2.0	2.5	1.8
1,1,1-TRICHLOROETHANE				7	7		
CARBON TETRACHLORIDE							
BENZENE		3.6		0.8	21	4.5	
1,2-DICHLOROETHANE							
TRICHLOROETHYLENE				12.0	27	32	39
1,2-DICHLOROPROPANE							
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE							
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE				3.6	3.0	<2.5	<2.5
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE							
ETHYL BENZENE							
m+p-XYLENE							
o-XYLENE							
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE							
TRIMETHYL BENZENE							
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	95	96	97	95	87	80	97

* SURROGATE was spiked at 100 µg/L.

Table A-2. Continued.

NATIONAL WATER RESEARCH INSTITUTE
GROUNDWATER CHEMISTRY LABORATORY
GC-MS ANALYSIS REPORT

DATE: 89/07/12

SAMPLE NUMBER	88GL67M09	88GL67M11	88GL82M06	88GL82M08	88GL92M15	88GL92M17	88GL92M19
CONCENTRATION	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							<6
VINYL CHLORIDE							<1
1,1-DICHLOROETHENE	4.5	37			7.3	18	16
DICHLOROMETHANE		61			7.3	11	4.1
DIETHYL ETHER	22	36			7.6	6.2	4.3
TETRAHYDROFURAN			20				
TRANS-1,2-DICHLOROETHENE					2.8	<1	2.1
1,1-DICHLOROETHANE	76	117			30	28	51
CIS-1,2-DICHLOROETHENE					<1	3.2	3.7
CHLOROFORM	2.8	137			147	164	32
1,1,1-TRICHLOROETHANE	42	520			87	87	100
CARBON TETRACHLORIDE							7.5
BENZENE	232	132			27	20	17
1,2-DICHLOROETHANE	19	58			22	16	9.9
TRICHLOROETHYLENE	325	583			82	74	28
1,2-DICHLOROPROPANE							3.2
DIBROMOMETHANE						<3	
TRANS-1,3-DICHLOROPROPENE							
TOLUENE		2.5					<0.7
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							<2
TETRACHLOROETHENE	<2.5	105					
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE		134			2.9	<2	
ETHYL BENZENE		2					
m+p-XYLENE							
o-XYLENE		10					
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE							
TRIMETHYL BENZENE							
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	85	74	99	75	86	85	96

* SURROGATE was spiked at 100 µg/L.

Table A-3. Compounds Tentatively Identified by GC/MS

SAMPLE	88GL126P	UNKNOWNS Dioxane approx. 330
SAMPLE	88GL16M12	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane 72
SAMPLE	88GL16M16	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane 23 dibromoethane dibromoethene
SAMPLE	88GL21M02	UNKNOWNS Dioxane approx. 330
SAMPLE	88GL21M03	UNKNOWNS Dioxane approx. 430
SAMPLE	88GL37M01	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane 20
SAMPLE	88GL37M06	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane 14 Dichlorotrifluoroethane Dichlorodifluoromethane
COMMENTS SAMPLE ANALYZED AT 2X DILUTION		
SAMPLE	88GL37M07	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane Diethyl disulfide Diisopropyl ether
COMMENTS SAMPLE ANALYZED AT 2X DILUTION		
SAMPLE	88GL37M09	UNKNOWNS Dichlorotrifluoroethane Dichlorodifluoromethane Diethyl disulfide Methyl ethyl sulfide Dimethyl sulfide 1,2-Dichloroethane
COMMENTS SAMPLE ANALYZED AT 5X DILUTION		

Table A-3. Continued.

SAMPLE	88GL47M01	UNKNOWNS Dioxane approx. 500 Unknown, mwt 132
SAMPLE	88GL47M04	UNKNOWNS Dioxane approx. 2000
SAMPLE	88GL54M06	UNKNOWNS Dichlorotrifluoroethane
SAMPLE	88GL67M01	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane 36
SAMPLE	88GL67M06	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane 40
SAMPLE	88GL67M07	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane 35
SAMPLE	88GL67M08	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane 50
COMMENTS SAMPLE ANALYZED AT 5X DILUTION		
SAMPLE	88GL67M09	UNKNOWNS 1,1,2-trichloro-1,2,2-trifluoroethane 140 Dichlorotrifluoroethane Dichlorodifluoromethane
COMMENTS SAMPLE ANALYZED AT 5X DILUTION		
SAMPLE	88GL67M11	UNKNOWNS Tetrachloroethane 15 1,1,2-trichloro-1,2,2-trifluoroethane 2725 Dichlorotrifluoroethane Chlorotrifluoroethene Dichlorodifluoromethane Diisopropyl ether Pentane Methoxybenzene Ethylcyclobutane
COMMENTS SAMPLE ANALYZED AT 5X DILUTION		

Table A-3. Continued.

SAMPLE	88GL92M15	**UNKNOWNS** Tetrachloroethane 6.9 1,1,2-trichloro-1,2,2-trifluoroethane 120 Dichlorotrifluoroethane Chloropropene
SAMPLE	88GL92M17	**UNKNOWNS** Tetrachloroethane ≈5 1,1,2-trichloro-1,2,2-trifluoroethane 115 Dichlorotrifluoroethane