

Ground-Water Contamination from Two Small Septic Systems on Sand Aquifers

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Abstract

Distinct plumes of septic system-impacted ground water at two single-family homes located on shallow unconfined sand aquifers in Ontario showed elevated levels of Cl^- , NO_3^- , Na^+ , Ca^{2+} , K^+ , alkalinity, and dissolved organic carbon and depressed levels of pH and dissolved oxygen. At the Cambridge site, in use 12 years, the plume had sharp lateral and vertical boundaries and was more than 130 m in length with a uniform width of about 10 m. As a result of low transverse dispersion in the aquifer, mobile plume solutes such as NO_3^- and Na^+ occurred at more than 50 percent of the source concentrations 130 m downgradient from the septic system. At the Muskoka site, in use three years, the plume also had discrete boundaries reflecting low transverse dispersion. After 1.5 years of system operation, the Muskoka plume began discharging to a river located 20 m from the tile field. Almost complete NO_3^- attenuation was observed within the last 2 m of the plume flowpath before discharge to the river. This was attributed to denitrification occurring within organic matter-enriched riverbed sediments.

The very weakly dispersive nature of the two aquifers was consistent with the results of recently reported natural-gradient tracer tests in sands. Therefore, for many unconfined sand aquifers, the minimum distance-to-well regulations for permitting septic systems in most parts of North America should not be expected to be adequately protective of well-water quality in situations where mobile contaminants such as NO_3^- are not attenuated by chemical or microbiological processes.

Introduction

About one-third of the population of the United States uses septic systems for waste-water disposal (U.S. EPA, 1986). Septic systems thus represent the largest volumetric source of effluent discharged to the ground-water zone. The literature, however, has few detailed field evaluations of septic system impacts on ground water (Childs et al., 1974; Walker et al., 1973b; Rea and Upchurch, 1980; Barber et al., 1988). Septic systems located on sand and gravel aquifers are a potential source for producing large-scale contaminant plumes in aquifers that are also likely to be used for drinking-water supply. Additional field evaluations are appropriate at this time for two reasons. First, recent studies have indicated that the dispersive capabilities, and therefore the contaminant dilution potential of many sand and gravel aquifers, are much less than previously thought (Sudicky et al., 1983; Freyberg, 1986; Garabedian, 1987; Molyaner and Killey, 1988a, b). This is significant because regulators frequently rely on dilution for attenuation of septic system contamination in ground water. Second, there has been recent concern regarding persistence of toxic trace organic constituents in the ground-water zone. Such contaminants frequently occur in sewage effluent (Viraraghavan and Hashem, 1986; Barber et al., 1988).

In this study, exceptionally detailed ground-water monitoring networks were used to investigate ground-water impacts caused by septic systems at two single-family homes located on shallow unconfined sand aquifers in Ontario. At the older site (Cambridge), where the septic system has been in operation since 1977 and where our field investigations began in 1987, a long narrow plume of septic system-impacted ground water, more than 130 m in length, has been identified within a carbonate-rich sand aquifer. At the younger site (Muskoka), a ground-water monitoring network was installed six months after the beginning of full-time use of the septic system in 1987, and monitoring was then carried out for a period of over two years. During that time, the plume migrated a distance of 20 m within a poorly buffered, carbonate-depleted, sand aquifer, and began discharging into an adjacent river. Both septic systems are of the conventional design used in Ontario and most other parts of North America for permeable soils. In this study the character of the plumes are described and major-ion geochemistry is evaluated along clearly defined subsurface flow paths originating from the septic system tile beds. In addition, because of the detail of the monitoring networks, we are able to infer dispersive characteristics of the aquifer at the Cambridge site and are able to evaluate the mobility of the major-ion species at the Muskoka site by monitoring plume breakthrough at downgradient locations. In subsequent papers the microbial conditions and the persistence and fate of a number of consumer product-derived organic constituents will be described in the Cambridge plume (Shimp et al., 1990; McAvoy et al., 1990; Robertson, 1990).

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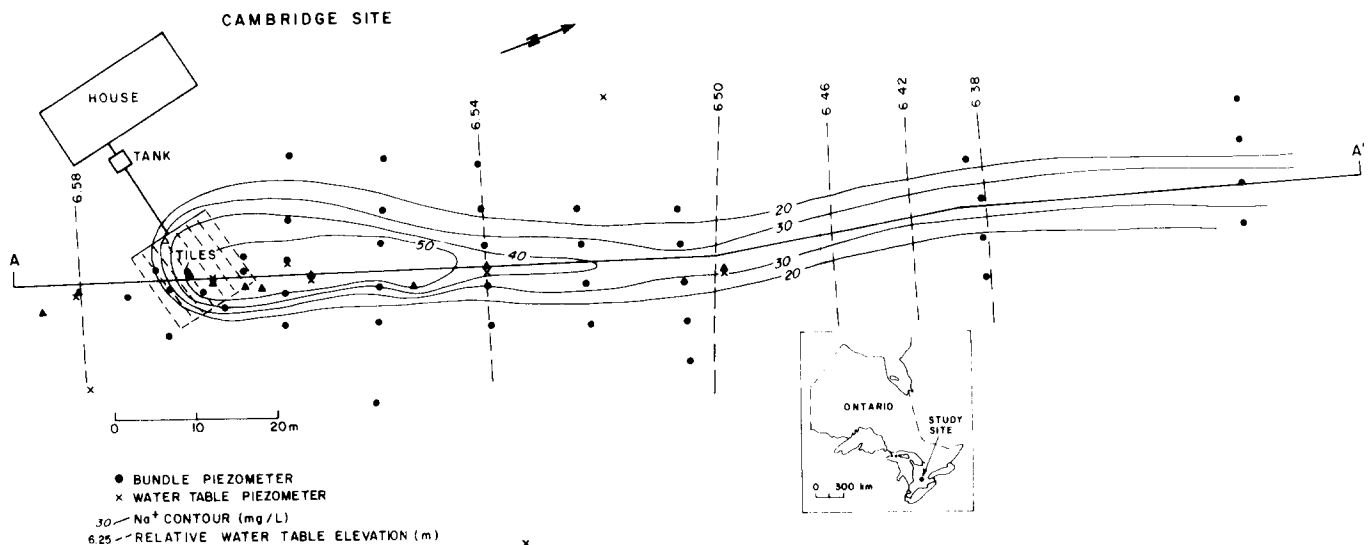


Fig. 1. Vertically averaged ground-water Na^+ distribution in the surficial aquifer, Cambridge site, 1987.

Field Sites

The Cambridge site is located near Cambridge, Ontario at an agricultural research station operated by the University of Guelph (Figure 1). The site is located on a flat-lying sand plain where glaciolacustrine and outwash sand occurs to a depth of 4-8 m and overlies a silt till of low permeability (Karrow, 1987). The surficial aquifer is comprised of moderately permeable fine sand to very permeable coarse sand with coarser material dominating. From 1977 to present, a family of two adults and two children have been permanent residents at the site. Since that time, household waste water including laundry effluent, has been discharged to a typical domestic septic system consisting of a holding tank and weeping tile bed 100 m² in area (Figure 1). The weeping tiles, consisting of perforated PVC pipe, are positioned in trenches 2 meters apart and are encased for a radius of 15 cm by gravel. The tiles lie at a depth of 0.6 m at a location where the water table is about 2 m below ground surface. The tile bed lies under a garden plot but most of the contaminant plume originating from the septic system extends under

intensively cultivated agricultural land to the northeast.

The Muskoka site is located on the edge of the Muskoka River near Bracebridge, Ontario (Figure 2). At this site, fine fluvial sand occurs to a depth in excess of 10 m and overlies granitic bedrock which outcrops within 100 m of the site. From 1987 to present, a family of two adults have lived at the site. Since that time household waste water, including laundry effluent, has been discharged to a typical domestic septic system consisting of a holding tank and tile bed about 80 m² in area. The tile bed is positioned 20 m from the Muskoka River at a location where the water table is about 3 m deep. Tile bed construction is the same as at Cambridge except that the tile lines are trenched into coarse sand fill that occurs to a depth of about 1 m. The tile bed and plume underlies grass lawn, and the upgradient ground-water flow system underlies uninhabited forested terrain.

Methods

The initial field investigation at each site involved installation of a network of water-table piezometers to establish the regional ground-water flow direction, the installation of a multiple piezometer bundle near the tile field, and the retrieval of a continuous 5-cm-diameter sediment core from the aquifer. The water-table piezometers were constructed of 1.6-cm-diameter PVC pipe with slotted and screened tips and were installed into the shallow water-table zone using a hand auger. The multiple piezometer bundles consisted of 13 2-mm-diameter polyethylene sampling tubes attached at 30-cm-depth intervals to a center stock of 1.6-cm-diameter PVC pipe. The bundles were installed into the aquifer with the aid of 5-cm-diameter steel casing and an expendable drive tip. The casing was advanced using a handheld vibrating hammer (Atlas Copco, COBRA) and was extracted after bundle insertion. Continuous sediment cores were obtained in 5-cm-diameter thin-walled aluminum core tubes using a newly developed portable coring apparatus (Starr and Ingleton, 1989). The core tubes, each 1.5 m in length, were fastened to the end of a 5-cm-diameter steel casing. Next, a plunger-type drive point

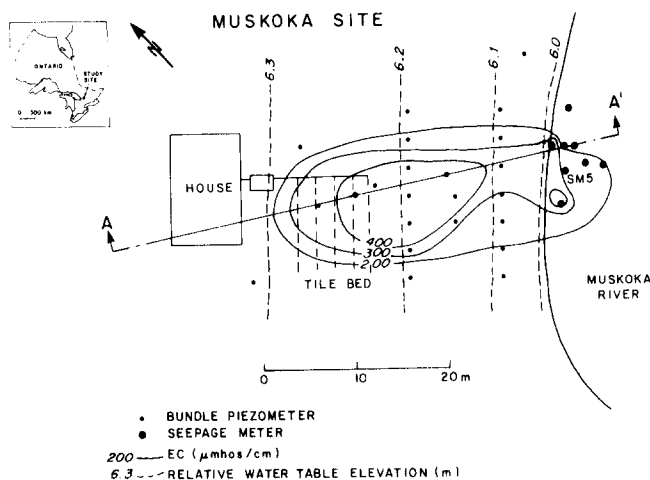


Fig. 2. Vertically averaged ground-water electrical conductivity at the Muskoka site, September 1988.

was inserted in the driving end of the core tube and was held in position with a steel center rod. The core was obtained by first advancing the coring apparatus to the desired depth using the handheld vibrating hammer. The plunger was then restricted from further movement using an anchored cable, and the core was obtained finally by advancing the core tube an additional 1.5 m. In excess of 90 percent core recovery was obtained using this technique.

After the preliminary field investigations, detailed ground-water monitoring networks were installed at each site. At Cambridge an additional 41 multiple piezometer bundles were inserted to the bottom of the aquifer at 4-6-m depth (Figure 1). At Muskoka, an additional 20 piezometer bundles were installed to depths of 3-5 m (Figure 2). Each piezometer bundle contained from 13 to 18 sampling tubes; hence the monitoring networks consisted of over 500 sampling points at Cambridge and over 250 sampling points at Muskoka.

After sampling and delineation of the plume locations in three dimensions, additional piezometer bundles were installed along the plume centerlines for the purpose of trace organics sampling. These bundles were each comprised of six Teflon tubes 0.64 cm in diameter arranged at 0.6-m-depth intervals. At Cambridge, nine such bundles were installed (piezometer bundles 33-41, Figure 3) while at Muskoka three were installed (piezometers 31-33, Figure 4).

During installation of each Teflon piezometer bundle, a continuous core of aquifer sediment was retrieved for the purpose of aquifer solids characterization.

At Muskoka, eight riverbed seepage meters similar to the type described by Lee (1977) were installed in the area of the riverbed where the plume was discharging. These were constructed from 0.3-m-diameter plastic buckets which were inverted and pressed into the sediment. These devices were successful in collecting discharging ground water which was sampled by means of a 0.6-cm-diameter access tube using a peristaltic pump.

Sampling access to the septic tank effluent was provided by means of a 0.6-cm-diameter sampling tube inserted into the center weeping tile at each site, from which effluent could be withdrawn using a peristaltic pump.

Routine sampling of the monitoring networks demonstrated that the zones of septic system-impacted ground water could be distinguished readily by field measurements of ground-water electrical conductance (EC) or by laboratory measurements of sodium (Na^+) levels. The entire monitoring networks were thus sampled regularly for electrical conductance using a temperature-corrected field-portable meter and a multiple-port sampling manifold, and about 50 percent of the monitoring network at Cambridge was also sampled for Na^+ content using the multiple-port sampling manifold or a peristaltic pump. Additional sampling at both sites was then concentrated along the plume centerlines and included sample collection for analysis of all major dissolved ions (Na^+ , K^+ , Ca^{2+} , Mg^{2+} , NH_4^+ , NO_3^- , Cl^- , SO_4^{2-} , and PO_4^{3-}), dissolved oxygen (DO), pH, alkalinity, and dissolved organic carbon (DOC). DO was measured in the field using the Winkler titration method (U.S. EPA, 1974) which provided a detection limit of about 0.1 mg/l. Alkalinity

was measured in the field by acid titration, and pH was measured in the field using a field-portable meter. Sodium analyses were performed using an atomic absorption method which provided a lower detection limit of 1 mg/l. Nitrate and bromide analyses were done by ion chromatography which provided a detection limit of 0.2 mg/l for NO_3^- -N and 0.5 mg/l for Br^- . Other major ion analyses were done colorimetrically.

Bromide tracer tests to establish effluent residence times in the septic tank, in the unsaturated zone, and in the ground-water zone at each site were conducted by instantaneously injecting a 1 kg mass of NaBr into the septic tanks. After injection, 20-ml samples were collected for analysis of Br^- content, from the tile effluent at Cambridge for a period of 11 days, and from the ground-water zone for a period of 350 days at Cambridge and 90 days at Muskoka.

Sediment grain-size distribution was determined using sieves for the sand fraction and a hydrometer for the silt and clay fractions. Total organic carbon fraction (foc) of the sediment was measured by combustion (Leco furnace) after acid leaching of the inorganic carbon. Sediment cation exchange capacity (CEC) was determined by a barium stripping technique (Peech et al., 1962). Sediment carbonate content (CaCO_3 equivalent) was determined by the method of Allison and Moodie (1965).

Falling head permeameter tests were conducted on repacked samples of core material to determine sediment hydraulic conductivity. Tests were done at 30-cm-depth intervals on each core using the apparatus and techniques described by Sudicky (1986).

Results

Stratigraphy

At the Cambridge site, sediment cores retrieved along section A-A' (piezometer bundle locations 33-51, Figures 1 and 3) showed that the surficial aquifer in the plume area is 4-6 m thick and overlies a slightly undulating till surface. Permeameter tests indicated that two major zones of contrasting hydraulic conductivity existed within the aquifer. In the area downgradient (north) of piezometer bundle 38, the entire thickness of the aquifer was comprised of clean medium to coarse sand (Table 1) having an average hydraulic conductivity (K) of 3×10^{-2} cm/s (sample size N = 21). In the area upgradient of bundle 38, including the area under the tile field, most of the saturated portion of the aquifer was comprised of less permeable fine- to medium-grained sand with average K of 5×10^{-3} cm/s (N = 22). The lower K zone was, however, underlain at all locations along the bottom of the aquifer by a high K layer of coarse sand approximately 0.3 m thick. Figure 5 shows a typical hydraulic conductivity profile for the tile bed area (core 36). The relatively impermeable underlying till unit consisted of compact pebbly sandy silt (K = 2×10^{-5} cm/s, N = 3). The contact between the aquifer and the till unit was generally a sharp erosional feature, except in the area between cores 35 and 38, where a topographic depression in the till surface was infilled with silt.

At the Muskoka site the zone below the water table consisted entirely of homogeneous silty fine sand (K = $1.3 \times$

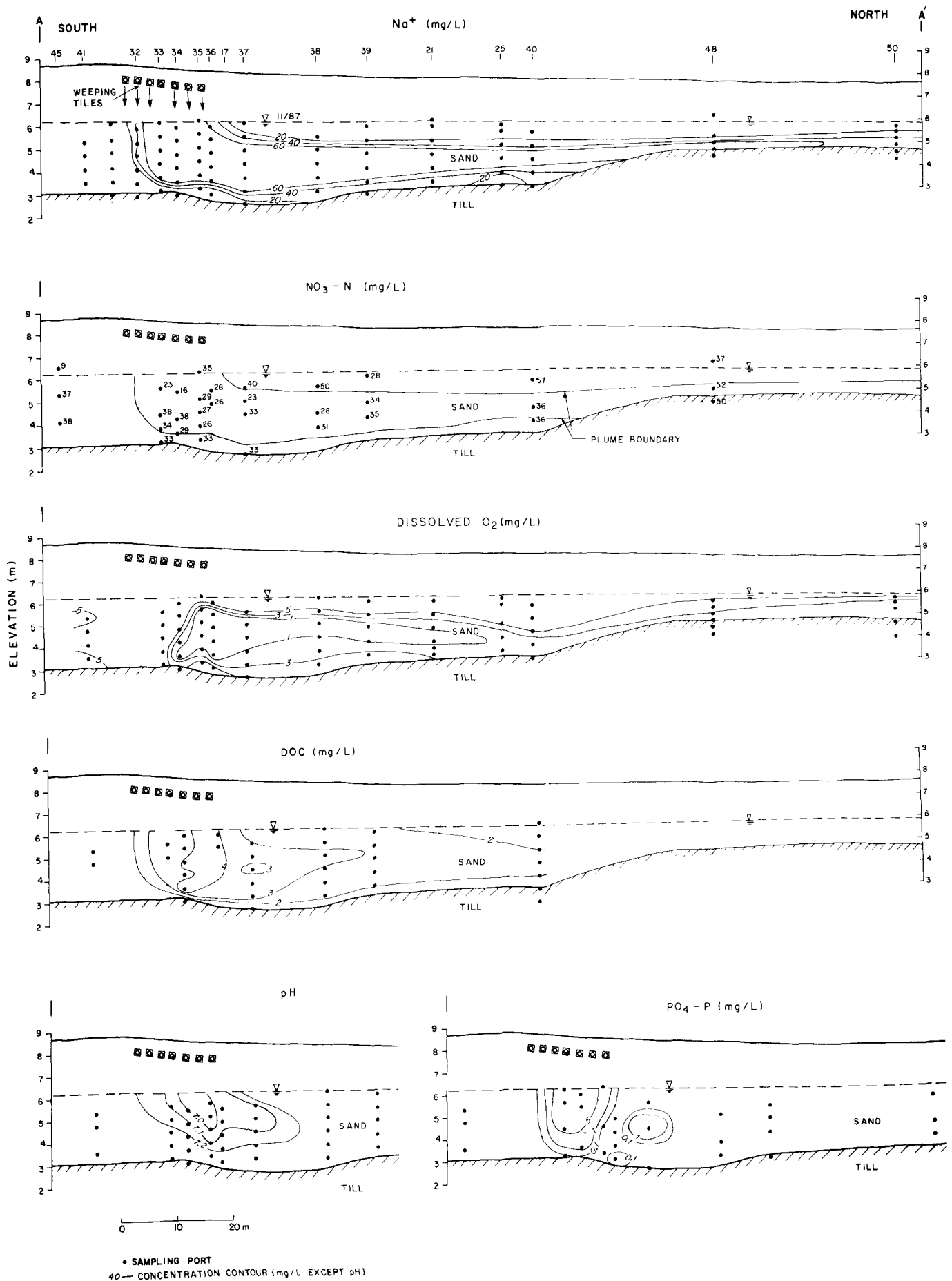


Fig. 3. Major ion geochemistry along the plume centerline at the Cambridge site, 1987-88.

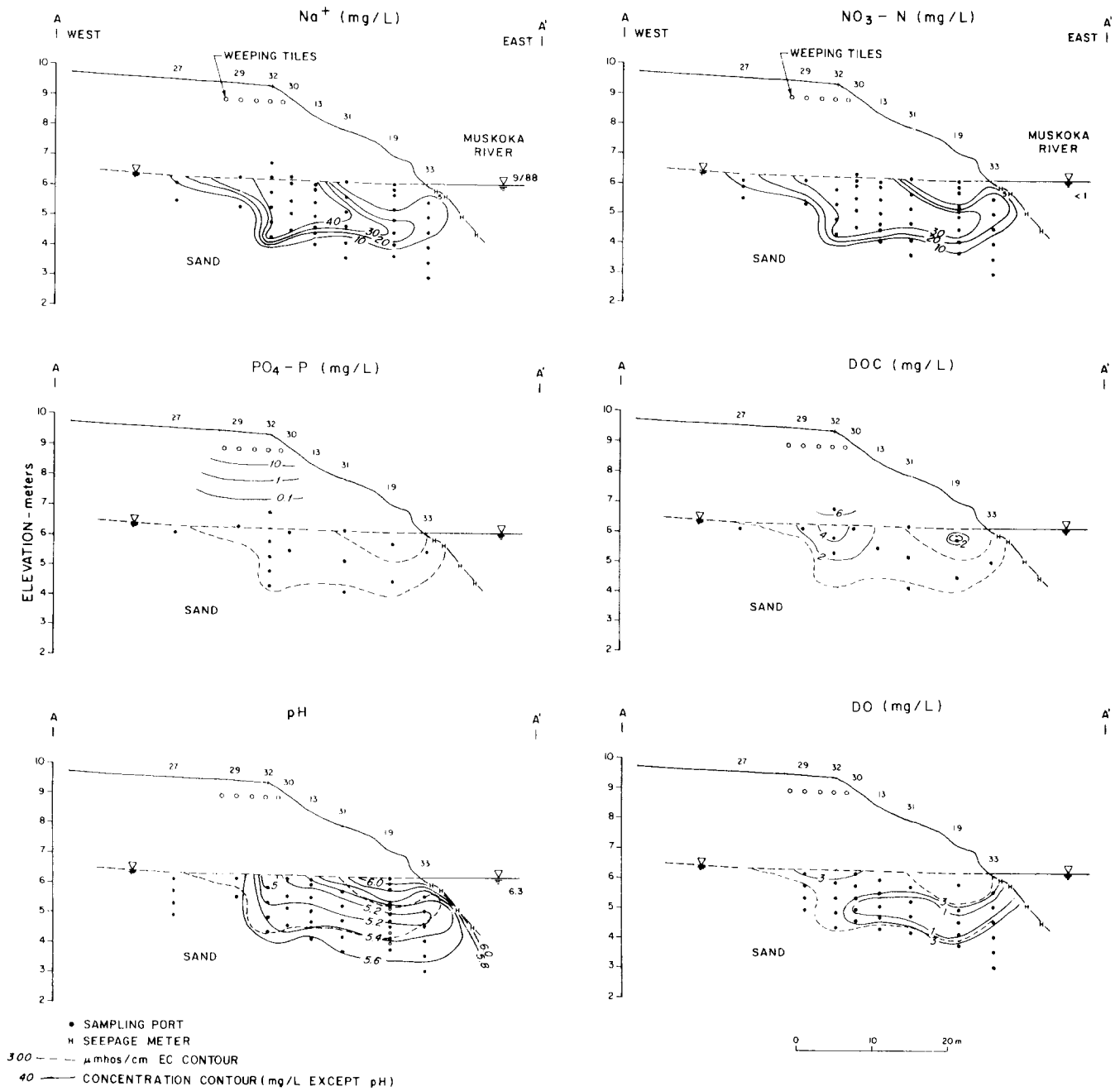


Fig. 4. Major ion geochemistry along the plume centerline at Muskoka, 500 days after the beginning of effluent discharge (September 1988).

Table 1. Aquifer Solids Properties

	Gravel (>2 mm) wt. %	Sand (>0.053 mm) wt. %	Silt wt. %	Clay (<0.002 mm) wt. %	Carbonate (CaCO_3 equiv.) wt. %	foc	CEC meq/100 g
Cambridge site Aquifer (N = 9)	0	95	5	0	21	0.0005 ¹	5.0
Muskoka site Aquifer (N = 7) Riverbed (N = 3) sediments	0	87	13	0	0.4	0.0003 ² 0.017 ³	2.2

¹ Sample size (N) = 7.

² N = 3.

³ 0.02 to 0.3 m below riverbed at seepage meter 5 location.

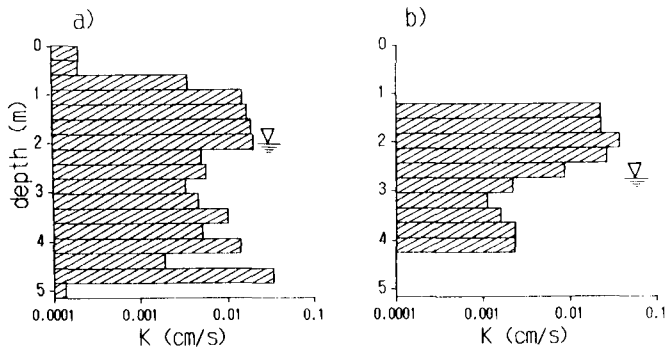


Fig. 5. Permeameter-derived hydraulic conductivity profiles below the tile bed areas: (a) Cambridge site, core 36, and (b) Muskoka site, core 32 (K at 10°C). Core locations are shown on Figures 3 and 4.

10^{-3} cm/s, $N = 38$) to a depth in excess of 7 m below the water table. The only contrasting sediment type observed at this site was a zone of clean medium sand ($K = 3 \times 10^{-2}$ cm/s, $N = 4$) occurring in the vadose zone below the tile bed (core 32, Figure 5) and a more silt-rich zone about 0.5 m thick on the riverbed.

Ground-Water Flow

At the Cambridge site, the water-table hydraulic gradient was northeasterly which was consistent with the plume-indicated direction of ground-water flow (Figure 1). The plume shape (Figure 3) demonstrated that flow within the aquifer was predominantly horizontal except beneath the tile bed where the plume had descended to near the bottom of the aquifer indicating a significant vertically downward component of flow. The Darcy equation was used to estimate the horizontal ground-water flow velocity in the downgradient area between piezometer bundles 39 and 40 (Figure 3). In this zone, the horizontal hydraulic gradient was about 0.0014 (Figure 1) and the permeameter-estimated hydraulic conductivity was about 3×10^{-2} cm/s. Assuming an effective porosity equal to 0.35, which is reasonable for this type of sand aquifer, the calculated flow velocity was about 40 m/a. In the area of the tile bed, lower flow velocities were expected because the horizontal hydraulic gradient was less, and K was generally lower. In this area, the Darcy equation was not used to estimate velocity because the hydraulic gradient was too low to measure with the required precision. The bromide tracer test, discussed in the next section, provided an indication of flow velocities in the immediate area of the tile field.

Bromide Tracer Tests

At the Cambridge site, the injected slug of NaBr resulted in maximum Br^- concentrations of 360 mg/l in the septic tank effluent and 89 mg/l in the ground-water zone below the tile field. These values were much higher than background Br^- concentrations (<0.5 mg/l). The test showed that the average septic tank residence time was about 2 days and that the effluent residence time in the 2-m-thick unsaturated zone and capillary fringe was on the order of 10 days. Figure 6 shows the position of peak Br^- concentrations in the ground-water zone at times of 10 to

350 days after injection. During the 350-day period, the slug migrated to a maximum depth of about 2.5 m below the water table and moved horizontally away from the tile bed a distance of 23 m downgradient. These data suggested a maximum vertical ground-water velocity of 1.7 cm/day in the area under the center of the tile bed and a horizontal velocity of 7 cm/day (24 m/a) in the area immediately downgradient of the tile field. Thus, the bromide tracer test and calculations using the Darcy equation suggested that the horizontal flow velocity in the surficial aquifer at Cambridge was on the order of 20 to 40 m/a.

At the Muskoka site, the bromide tracer test was less successful because there were fewer monitoring points and these were sampled less frequently. It appears that the Br^- tracer experienced a longer residence time, on the order of several weeks to months, in the 3-m-thick unsaturated zone at this site. The longer residency at Muskoka was likely the result of lower effluent dose rates (only two persons using the system) and higher evapotranspirative losses during the test period. The tracer test was begun in late June at Muskoka while it was begun in early May at Cambridge. Elevated Br^- concentrations were observed under the tile bed, however, at a depth of 0.5 m below the water table, 90 days after injection. Although the bromide test was unsuccessful, an alternative method provided an estimate for ground-water velocity. The velocity along most of the septic system plume was determined by observing the rate of advance of the zone of elevated EC (Figure 6). This was possible because ground water impacted by the tile effluent was easily identified by its contrasting electrical conduc-

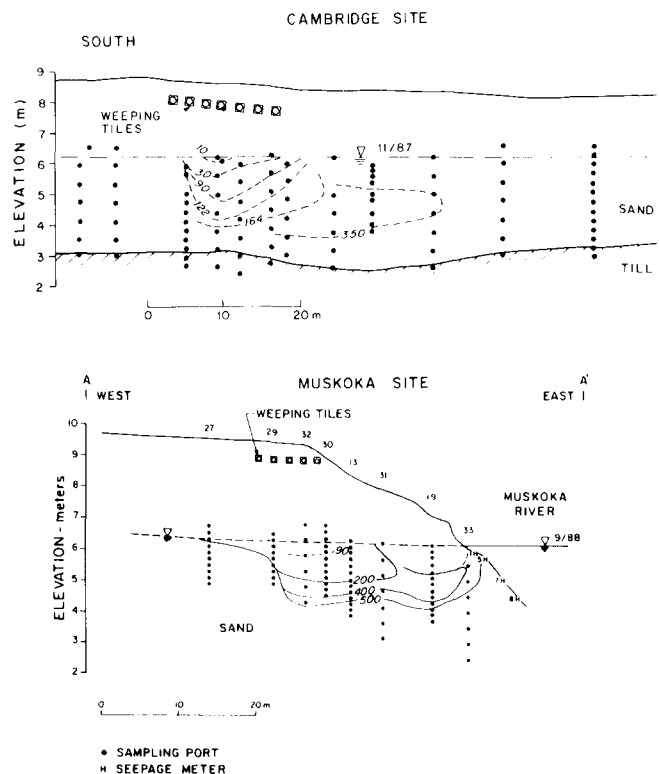


Fig. 6. Ground-water residence time (days) along the plume centerlines, inferred from bromide tracer tests (dashed lines) and from breakthrough of $300 \mu\text{mhos/cm}$ EC front (solid lines).

tance. A horizontal flow velocity of about 20 m/a was indicated by this method.

At the Muskoka site, there is a large water-table gradient of about 0.01 in a direction toward the river (Figure 2). Using the permeameter-indicated average K value of 1.3×10^{-3} cm/s and assuming a porosity of 0.35, the Darcy equation gave a horizontal flow velocity of 12 m/a. Thus, calculations using the Darcy equation and observations of plume front arrival suggested that horizontal flow rate was on the order of 10 to 20 m/a at the Muskoka site.

Plume Character

Table 2 summarizes the major-ion geochemistry of the background ground water, the tile effluent, and the plume core ground water at both sites. Figure 1 shows vertically averaged Na^+ distribution in the surficial aquifer at Cambridge, while Figure 3 shows Na^+ distribution along the plume centerline (section A-A'). At the Cambridge site, plume core water was characterized by high Na^+ (71 to 91 mg/l), Cl^- (21 to 29 mg/l), and NO_3^- -N (21 to 48 mg/l). However, because background ground water at this site also had high Cl^- (10 to 29 mg/l) and NO_3^- -N (17 to 34 mg/l), Na^+ was found to be the best indicator of impact from the septic system because it occurred in the plume at 10 to 20 times the background level. The high background NO_3^- and Cl^- values were the result of agricultural practices such as manure and chemical fertilizer application.

A long, narrow plume more than 130 m long and about 10 m wide was defined at Cambridge (Figure 1). Sodium concentrations that were more than 50 percent of the effluent value occurred 130 m downgradient from the tile field. This demonstrated the relatively rapid advection of the plume and the ineffectiveness of dispersion, laterally and vertically, to significantly attenuate the plume. Figure 3 shows that in the area of the tile field, the septic system plume was about 2.5 m thick and occupied the entire thickness of the aquifer except for the bottom 0.5 m. Downgra-

dient from the tile field, the plume became overlain by an increasing thickness of nonimpacted ground water recharged from the overlying farm field. The boundary between the plume and the overlying nonimpacted zone was abrupt which reflects the low vertical transverse dispersion capability of the aquifer material.

Figure 3 also shows DO, NO_3^- -N, PO_4^{3-} -P, DOC, and pH distribution along section A-A'. The plume core was characterized by depressed levels of DO (<1 mg/l) compared to background values of 3 to 7 mg/l, but detectable amounts of DO (>0.1 mg/l) persisted at all locations. NO_3^- -N levels were two to four times in excess of the Ontario drinking-water limit of 10 mg/l (MOE, 1983). PO_4^{3-} was observed in the plume only in the area beneath the tile field. The plume core showed slightly depressed pH levels (7.0) compared to background values (7.3).

Major ion geochemistry along the plume core at Muskoka (Figure 4) was similar to that at Cambridge for some species, with comparable levels of Na^+ , Cl^- , and NO_3^- -N measured. The plume descended to a depth 2 m below the water table in the tile area and became overlain downgradient by nonimpacted ground water recharged through the overlying lawn area. The upper and lower plume boundaries were again sharp, demonstrating the low vertical dispersion capability of the aquifer and again demonstrating the ineffectiveness of dilution in attenuating plume solute concentrations. Depressed levels of DO were measured along the plume core with concentrations similar to those at Cambridge observed. The plume core again exhibited depressed levels of pH (5.1) compared to background values (5.7). There was no detectable PO_4^{3-} (>0.02 mg/l PO_4^{3-} -P) in the ground-water zone at Muskoka.

Nitrogen in septic system effluent occurs about 80 percent as inorganic nitrogen, with NH_4^+ -N dominating (Walker et al., 1973a; Andreoli et al., 1979). Inorganic nitrogen commonly occurs at concentrations ranging from 30-111 mg/l in septic system effluent (Walker et al., 1973a;

Table 2. Major Ion Geochemistry

	Cambridge site			Muskoka Site		
	Upgrad. N = 3	Tile effluent ¹ N = 4	Plume ² core N = 7	Upgrad. N = 2	Tile effluent ¹ N = 5	Plume ³ core N = 8
Na, mg/l	4	98	86	2	90	45
K, mg/l	2	12	11	1	21	14
Ca, mg/l	88	40	90	8	14	44
Mg, mg/l	16	14	17	1	3	3
Alk, mg/l	204	365	276	7	316	12
SO_4 , mg/l	59	27	63	6	42	32
Cl , mg/l	17	45	24	4	55	38
NO_3 -N, mg/l	27	1	33	3	0.1	39
NH_4 -N, mg/l	<0.1	30	0.1	<0.1	59	0.5
PO_4 -P, mg/l	<0.01	8	4	<0.01	13	0.01
DOC, mg/l	3	37	4	1.6	50	3.4
pH	7.3	7.9	7.0	5.7	7.6	5.1
EC, $\mu\text{mhos/cm}$	770	1020	940	75	950	550

¹ Average composition of samples from weeping tile access tube.

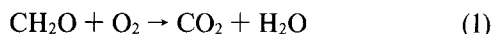
² Average composition of samples from piezometer bundles; 33, 3.0 m and 4.2 m depths; 34, 2.4 m, 3.0 m, and 4.8 m depths; and 35, 2.1 m and 3.8 m depths. Locations shown on Figure 3.

³ Average composition of samples from piezometer bundles; 32, 3.5 m depth; and 30, 2.6 m depth. Locations shown on Figure 4.

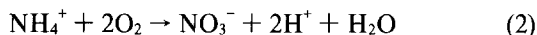
Viraraghaven and Warnock, 1975; Andreoli et al., 1979). Thus, the total inorganic nitrogen concentrations of the effluent at Cambridge and Muskoka ($\text{NH}_4^+\text{-N} + \text{NO}_3^-\text{-N}$ from Table 2; 31 and 59 mg/l, respectively) appear typical of most septic system effluents. It is difficult to find literature values for nitrogen content in septic system-impacted ground waters from sandy aerobic environments, however, several workers report $\text{NO}_3^-\text{-N}$ concentrations in the subsurface below tile lines that are similar to our ground-water values (Walker et al., 1973b; Andreoli et al., 1979; 10-50 mg/l $\text{NO}_3^-\text{-N}$). $\text{NO}_3^-\text{-N}$ values in septic system-impacted ground waters are commonly above the drinking-water limit of 10 mg/l (Childs et al., 1974; Rea and Upchurch, 1980). Thus, our NO_3^- results and plume distances are probably not unusual.

Geochemical Processes

A comparison of tile effluent and plume core chemistry (Table 2) shows that at both sites plume water contained much less DOC (1.6-3 vs 37-50 mg/l), much higher $\text{NO}_3^-\text{-N}$ (33-39 vs 0.1-1 mg/l), and much lower $\text{NH}_4^+\text{-N}$ (0.1-0.5 vs 30-59 mg/l) than the tile effluent. This suggested that: (1) organic carbon in the effluent was being consumed by an aerobic biodegradation reaction such as;



where the simple carbohydrate CH_2O is used conceptually here to represent organic matter, and that (2) ammonium in the effluent was being oxidized via microbial nitrification,



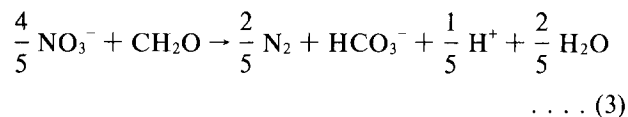
The low DOC levels and high NO_3^- levels observed in even the shallowest water-table zone below the tile fields indicates that these processes were completed largely during the residency of the effluent in the unsaturated zone. Likewise, Walker et al. (1973a) observed significant nitrification of NH_4^+ within 6 cm below the gravel pack/soil interface and after effluent residency of only a few hours, in sandy aerobic unsaturated zones below several septic systems in Wisconsin. However, the continued depletion of DO and DOC in the zone below the water table (Figures 3 and 4) suggests that these processes continued to some extent below the water table. Since ammonium-N found in the tile effluent (30-59 mg/l) was approximately equivalent to nitrate-N found in the plumes (33-39 mg/l), the tile effluent was the probable source of nearly all of the nitrate in the plumes. A similar conclusion was reached by Walker et al. (1973a) regarding the source of pore-water NO_3^- in the unsaturated zones below the septic systems in Wisconsin. The relatively long residency (>7 days) of the effluent in the aerobic unsaturated zones at both Cambridge and Muskoka allowed reactions (1) and (2) to be completed largely within the vadose zones at these sites. At other locations where vadose zone residency is shorter due to a shallow water-table condition or due to a higher effluent infiltration rate, these reactions may not be complete within the vadose zone. This would tend to cause higher NH_4^+ , lower NO_3^- , higher DOC, and lower DO, as well as other differences in plume chemistry.

Acidity produced by NH_4^+ nitrification [equation (2)]

has resulted in depressed pH levels in the cores of both plumes (Figures 3 and 4; Table 2). At Cambridge, the pH depression was minor compared to background pH (7.0 versus 7.3) due to carbonate buffering, but at Muskoka, it was more significant (5.1 versus 5.7) due to low buffering capacity of the carbonate-poor aquifer sands there (Table 1). At Muskoka, pH levels in the plume may be sufficiently low to cause enhanced mobility of some constituents such as metals. This topic will be investigated in a subsequent study.

At Muskoka, plume development and migration rate has been documented because ground-water monitoring was begun six months after the beginning of system operation. Figure 7 shows normalized breakthrough of the major ions at piezometer 19-9 located in the plume core 17 m downgradient of the tile bed, and at seepage meter 5 located on the riverbed in the center of the plume discharge area (Figure 2). Source concentrations for normalization (C_0) were taken as the plume core average values given in Table 2. At piezometer 19-9, NO_3^- and Cl^- appeared most mobile, arriving after about 300 days of system operation, at about the same time that elevated EC levels were observed. Na^+ arrival was slightly retarded compared to Cl^- , indicating probable minor adsorption of Na^+ . Na^+ remained relatively mobile, however, and had a migration velocity at least 75 percent that of Cl^- . K^+ was more highly attenuated and retarded than Na^+ . Table 1 indicates that the Muskoka aquifer sand has a low cation exchange capacity (2 meq/100 g).

The condition of NO_3^- breakthrough in the riverbed seepage meter is of particular interest at 500 to 600 days of system operation. $\text{NO}_3^-\text{-N}$ levels remained < 0.5 mg/l in the seepage meter while $\text{Cl}^- C/C_0$ values rose to 0.3 (Figure 7b). This was observed even though NO_3^- occurred at uniformly high concentrations and was shown to be very mobile throughout the remainder of the plume (Figure 7a). It is likely that vigorous denitrification occurred in the riverbed sediments as a result of the development of anaerobic conditions there. In such an environment, NO_3^- can be converted to nitrogen gas by a reaction such as,



where CH_2O represents organic matter. Such a phenomenon is strongly suggested by Figure 4 and Table 3 which show that $\text{NO}_3^-\text{-N}$ decreased from about 20 mg/l to less than 0.5 mg/l in the last meter or so of the flow path before discharging into the river. Starr (1988) demonstrated the role of labile organic carbon in promoting the development of anaerobic conditions and denitrification in sandy aquifers. Increased availability of organic carbon in the riverbed sediments is suggested from Table 1, which shows an floc level, for the riverbed sediment, 60 times higher than that of the aquifer sand. Table 3 indicates that the most vigorous zone of denitrification was at a depth of 0.5 m below the riverbed. Nitrate continued to be attenuated in the riverbed sediment at Muskoka; however, some variability in the attenuation capacity, perhaps seasonally or water-level-related, was indicated by the temporary breakthrough of NO_3^- during the July 1989 sampling ($T = 840$ days, Figure 7b).

Table 3. NO_3^- , Cl^- , and NH_4^+ Concentrations in Riverbed Sediments, Muskoka Site; Ground-Water Samples Obtained from Seepage Meter 5 and from Drive-Point Piezometers Installed Below Seepage Meter 5

Depth below riverbed (m)	Sampling date	NO_3^- -N mg/l	Cl mg/l	NH_4^+ -N mg/l	Sampling date	NO_3^- -N mg/l	Cl mg/l	NH_4^+ -N mg/l
0.0 (SM5)	12/88	0.6	14	0.3	10/89	<0.05	22	<0.05
0.25	4/89	3.1	15	<0.05	10/89	<0.05	36	0.24
0.50	4/89	13	7	<0.05	10/89	8.6	30	<0.05
river water	9/88	0.1	1.9	<0.05				

Phosphate occurred in the effluent at both sites at concentrations of about 10 mg/l PO_4^{3-} -P (Table 2), but was highly attenuated in the subsurface. Such a condition is commonly observed for PO_4^{3-} in subtile sediments (Sawhney and Starr, 1977; Jones and Lee, 1979). At Cambridge, PO_4^{3-} -P levels of > 1 mg/l were mostly confined to the aquifer area immediately below the tile bed, while at Muskoka, no detectable PO_4^{3-} -P (>0.02 mg/l) was observed in the ground-water zone. Although the phosphate attenuation processes have not as yet been rigorously evaluated, at Cambridge it was noted that very little phosphate attenuation occurred in the unsaturated zone while significant attenuation (>5 mg/l to <0.02 mg/l) occurred after several meters of flow within the ground-water zone. The high-phosphate ground waters have been shown to be oversaturated with respect to hydroxylapatite [$\text{Ca}_5(\text{OH})(\text{PO}_4)_3$] (Wilhelm et al., 1990), and the zone of phosphate attenuation is also a zone of increasing pH (7.0 to 7.3) and increasing Ca^{2+} concentration (80 to 100 mg/l) which are factors that would decrease hydroxylapatite solubility. Phosphate attenuation

at Cambridge may thus be controlled by hydroxylapatite precipitation. At the Muskoka site, PO_4^{3-} mobility was less likely controlled by hydroxylapatite solubility because Ca^{2+} concentrations in the plume were lower (44 mg/l), and pH was much lower (5.1). Phosphate at Muskoka may be attenuated by the precipitation of other sparsely soluble phosphate minerals such as strengite ($\text{FePO}_4 \cdot 2\text{H}_2\text{O}$), or varisite ($\text{AlPO}_4 \cdot 2\text{H}_2\text{O}$) or may be controlled by sorption.

Aquifer Dispersion

At Cambridge, a three-dimensional analytical model (Sudicky, 1985) was employed to obtain preliminary estimates of the aquifer dispersion parameters within the saturated zone. For the simulations, a vertical patch source 1.75 m thick by 8 m wide in the direction perpendicular to flow, was positioned in the middle of the saturated part of the aquifer at the downgradient edge of the tile field. Aquifer thickness was set at 3 m. A uniform horizontal flowfield with average linear ground-water velocity of 30 m/a was assigned to the aquifer; otherwise, no boundary fluxes were specified. To simulate Na^+ distribution, ground water exiting the source was assigned a solute concentration of 80 mg/l. Na^+ was assumed to be nonreactive. Figure 8 shows that Na^+ concentrations along the plume core were well represented by the model when low dispersion parameters typical of values determined from recent natural-gradient tracer experiments in sands (Sudicky et al., 1983; Freyberg, 1986; Garabedian, 1987; Moltyaner and Killey, 1988a, b) are used [i.e., longitudinal dispersivity (α_L) = 1 m, vertical transverse dispersivity (α_{tv}) = 0.004 m, and horizontal transverse dispersivity (α_{th}) = 0.01 m]. Use of larger values of α_{tv} or α_{th} resulted in Na^+ concentrations along the plume core that were less than those observed (Figure 8). An effective diffusion coefficient (D^*) of 0.02 m^2/a was used for all simula-

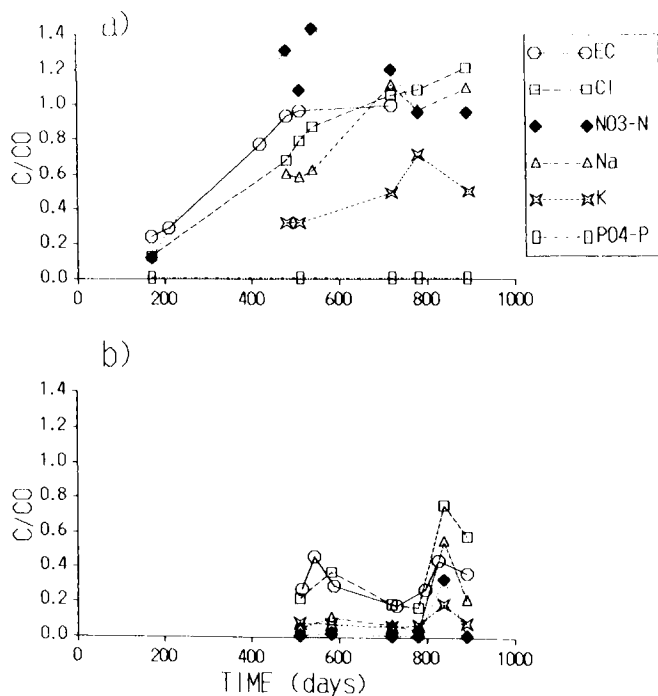


Fig. 7. Normalized major ion breakthrough at the Muskoka site: (a) 17 m downgradient from the tile field (piezometer 19, 2.4 m depth), and (b) 24 m downgradient (seepage meter 5). (t_0 = April 1987, x_0 = center of tile field, C_0 from plume core values given in Table 2, except effluent value used for PO_4^{3-} -P).

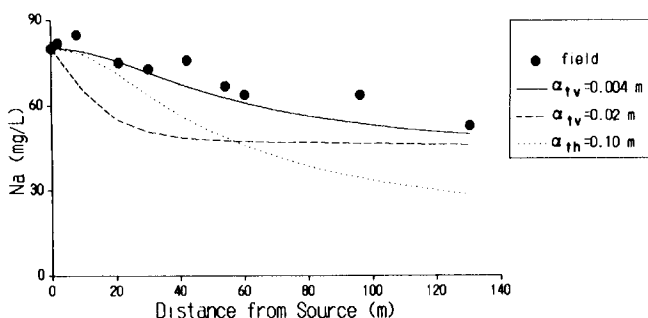


Fig. 8. Simulated steady-state Na^+ distribution along the plume core at Cambridge using a 3D analytical model ($\alpha_L = 1$ m, $\alpha_{th} = 0.01$ m, $\alpha_{tv} = 0.004$ m except where noted, $v = 30$ m/a, $D^* = 0.02$ m^2/a).

tions. The closeness with which observed Na^+ concentrations could be simulated using a constant source concentration of 80 mg/l suggested that Na^+ loading from the septic system had been consistent during the operation of the system. This was also suggested by Na^+ concentrations observed in the plume core below the tile bed where, from 19 sampling points representing effluent discharged over about a six-month period, all Na^+ values were between 71 and 91 mg/l.

Figure 9, which shows simulated vertical Na^+ profiles 30 m downgradient from the tile field (piezometer bundle 39), demonstrates that a very low α_{tv} value of about 0.004 m or less was necessary to maintain the sharp Na^+ gradient along the plume upper boundary. Figure 10 demonstrates that a low value of α_{th} on the order of a few centimeters or less was necessary to maintain the narrow plume width (10 m) at downgradient locations. These simulations thus confirmed our observation that the aquifer at Cambridge had a low capacity for dispersion. Although no simulations of the Muskoka plume were attempted, the sharp plume boundaries suggest that dispersion there was also low.

Conclusions and Implications

Modeling results indicated that the vertical transverse dispersivity of the Cambridge aquifer was on the order of 0.4 cm or less and that horizontal transverse dispersivity was less than 10 cm. Dispersion in the Muskoka aquifer also appeared to be low. The weak transverse dispersion process observed in the two plumes is significant in that it is consistent with very detailed tracer tests of the type recently performed at Borden and Twin Lakes, Ontario, and at Cape Cod, Massachusetts.

It is of interest to examine the implication of these dispersion values with respect to the impact of septic systems on ground water. Although no field estimates for longitudinal dispersion were obtained from this study, sensitivity analyses conducted using the previously discussed analytical model demonstrated that this parameter is likely to be unimportant in diluting plumes in low dispersion sands, even over long distances. For example, for a Cambridge-type plume 300 m in length, the portion of the plume

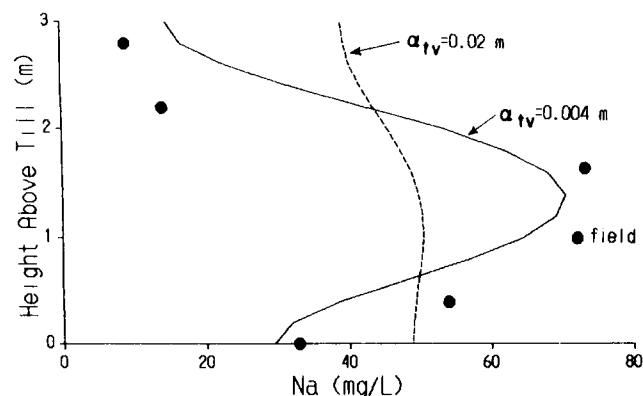


Fig. 9. Simulated steady-state vertical Na^+ distribution 30 m downgradient from the tile field (piezometer bundle 39) at the Cambridge site using a 3D analytical model ($\alpha_L = 1$ m, $\alpha_{th} = 0.01$ m, $v = 30$ m/a, $D^* = 0.02$ m²/a).

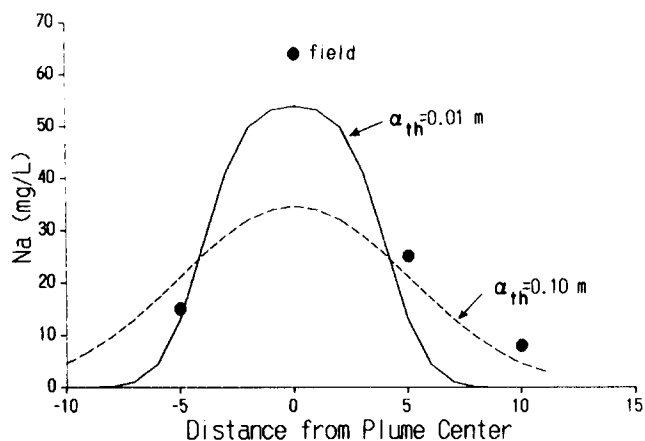


Fig. 10. Simulated steady-state Na^+ distribution transverse (horizontally) to the plume core at Cambridge, 95 m downgradient from the tile field ($v = 30$ m/a, $\alpha_L = 1$ m, $\alpha_{tv} = 0.004$ m, $D^* = 0.02$ m²/a).

influenced by longitudinal dispersion would be only the frontal 50 m or so when α_L was equal to 1 m. For the remainder of the plume length, dilution of nonreactive solutes would occur by transverse dispersion only. This conclusion is consistent with model sensitivity studies conducted by Frind and Hokkanen (1987). Using the analytical model and again considering the Cambridge example, the portion of the steady-state plume that would have contaminants at concentrations above drinking-water limits was estimated. For a plume with a source concentration of 33 mg/l NO_3^- -N, and where NO_3^- was absent in background ground water, the steady-state plume length in which NO_3^- -N would be in excess of the drinking-water limit of 10 mg/l is 170 m when α_{th} and α_{tv} values are 0.1 m and 0.004 m, respectively. However, the plume length in which NO_3^- -N would be in excess of one-fourth of the drinking-water limit (2.5 mg/l) is much longer, about 2 km. In some jurisdictions (i.e., Province of Ontario) this is the allowable impact from a single septic system. These "above the limit" plume lengths are estimated using the upper range of the dispersion parameters indicated by the field data at Cambridge. Even longer plumes result if smaller dispersion parameters are considered for sandy aquifers. Such examples demonstrate that for many unconfined sand aquifers, the typical minimum permissible distance-to-wells regulations that exist in most parts of the United States and Canada for septic systems (25-35 m) should not be expected to be adequately protective of well-water quality, except in those circumstances where it can be shown that significant mobile contaminants such as NO_3^- are attenuated by processes other than dispersion (i.e., by biodegradation). An example of such attenuation is provided by the Muskoka site, where the NO_3^- plume is terminated by denitrification.

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References

- Allison, L. E. and C. D. Moodie. 1965. Carbonate. In: Methods of Soils Analysis, Part 2. Black, C. A., ed. pp. 1379-1396.
- Andreoli, A., N. Bartilucci, R. Forgione, and R. Reynolds. 1979. Nitrogen removal in a subsurface disposal system. *J. Wat. Pollut. Control Fed.* v. 51, pp. 841-854.
- Barber, L. B., II, E. M. Thurman, M. P. Schroeder, and D. R. LeBlanc. 1988. Long-term fate of organic micropollutants in sewage-contaminated groundwater. *Environ. Sci. Technol.* v. 22, pp. 205-211.
- Childs, K. E., S. B. Upchurch, and B. Ellis. 1974. Sampling of variable, waste-migration patterns in ground water. *Ground Water.* v. 12, pp. 369-376.
- Freeze, R. A. and J. A. Cherry. 1979. *Groundwater*. Prentice-Hall, Inc., Englewood Cliffs, NJ.
- Freyberg, D. L. 1986. A natural gradient experiment on solute transport in a sand aquifer, 2. Spatial moments and the advection and dispersion of nonreactive tracers. *Water Resour. Res.* v. 22, pp. 2031-2046.
- Frind, E. O. and G. E. Hokkanen. 1987. Simulation of the Borden plume using the alternating direction Galerkin technique. *Water Resour. Res.* v. 23, pp. 918-930.
- Garabedian, S. P. 1987. Large-scale dispersive transport in aquifers: Field experiments and reactive transport theory. Ph.D. thesis, Mass. Inst. of Tech. 290 pp.
- Jones, R. A. and G. F. Lee. 1979. Septic tank wastewater disposal systems as phosphorus sources for surface water. *J. WPCF.* v. 51, pp. 2764-2775.
- Karrow, P. F. 1987. Quaternary geology of the Hamilton-Cambridge area, southern Ontario. Ontario Geological Survey Report 255. 94 pp.
- Lee, D. R. 1977. A device for measuring seepage of flux in lakes and estuaries. *Limnol. and Oceanog.* v. 22, pp. 140-147.
- McAvoy, D. C., C. E. White, and R. A. Rapaport. 1990. A case study of groundwater contamination from a domestic septic system: 4. Sorption and transport of anionic and cationic surfactants. *Env. Tox. and Chem.* In prep.
- MOE (Ontario Ministry of the Environment). 1983. Ontario drinking water objectives. Technical report. 56 pp.
- Moltyaner, G. L. and R.W.D. Killey. 1988a. Twin Lake tracer tests: Longitudinal dispersion. *Water Resour. Res.* v. 24, pp. 1613-1627.
- Moltyaner, G. L. and R.W.D. Killey. 1988b. Twin Lake tracer tests: Transverse dispersion. *Water Resour. Res.* v. 24, pp. 1628-1637.
- Peech, M., R. L. Cowan, and J. H. Baker. 1962. A critical study of the BaCl_2 triethanolamine and ammonium acetate methods for determining the exchangeable hydrogen content of soils. *Soil Sci. Soc. Am.* v. 26, pp. 37-40.
- Rea, R. A. and S. B. Upchurch. 1980. Influence of regolith properties on migration of septic tank effluent. *Ground Water.* v. 18, pp. 118-125.
- Robertson, W. D. 1990. A case study of groundwater contamination from a domestic septic system: 7. Persistence of dichlorobenzene. *Env. Tox. and Chem.* In prep.
- Sawhney, B. L. and J. L. Starr. 1977. Movement of phosphorous from a septic system drainfield. *J. WPCF.* v. 49, pp. 2238-2242.
- Shimp, R. J., E. V. Lapsins, and R. N. Ventullo. 1990. A case study of groundwater contamination from a domestic septic system: 6. Biodegradation of linearalkylbenzenesulfonate and nitrotriacetic acid. *Env. Tox. and Chem.* In prep.
- Starr, R. C. 1988. An investigation of the role of labile organic carbon in denitrification in shallow sandy aquifers. Ph.D. thesis, Univ. of Waterloo. 148 pp.
- Starr, R. C. and R. A. Ingleton. 1989. A new method for collecting core samples without a drill rig. Technical report, Institute for Groundwater Research, Univ. of Waterloo. 18 pp.
- Sudicky, E. A., J. A. Cherry, and E. O. Frind. 1983. Migration of contaminants in ground water at a landfill: A case study, 4. A natural gradient dispersion test. *J. Hydrol.* v. 63, pp. 81-108.
- Sudicky, E. A. 1985. A collection of analytical solutions for solute transport in porous and fractured media. Technical report, Institute for Groundwater Research, Univ. of Waterloo, Ontario.
- Sudicky, E. A. 1986. A natural-gradient experiment on solute transport in a sand aquifer: Spatial variability of hydraulic conductivity and its role in the dispersion process. *Water Resour. Res.* v. 22, pp. 2069-2082.
- U.S. EPA (Environmental Protection Agency). 1974. Methods for chemical analysis of water and wastes. Technical report, Office of Technology Transfer, Washington, DC. pp. 105-106.
- U.S. EPA (Environmental Protection Agency). 1986. Septic systems and ground water protection: An executives' guide. Technical report, Office of Groundwater Protection, Washington, DC.
- Viraraghavan, T. and S. Hashem. 1986. Trace organics in septic tank effluent. *Water, Air, and Soil Pollut.* v. 28, pp. 299-308.
- Viraraghavan, T. and R. G. Warnock. 1975. Groundwater pollution from a septic tile field. *Water, Air, and Soil Pollut.* v. 5, pp. 281-287.
- Walker, W. G., J. Bouma, D. R. Keeney, and F. R. Magdoff. 1973a. Nitrogen transformation during sursurface disposal of septic tank effluent in sands: I. Soil transformations. *J. Environ. Quality.* v. 2, pp. 475-480.
- Walker, W. G., J. Bouma, D. R. Keeney, and P. G. Olcott. 1973b. Nitrogen transformation during sursurface disposal of septic tank effluent in sands: II. Ground water quality. *J. Environ. Quality.* v. 2, pp. 521-525.
- Wilhelm, S. R., S. L. Schiff, W. D. Robertson, and J. A. Cherry. 1990. A case study of groundwater contamination from a domestic septic system: 3. Unsaturated and saturated zone geochemistry. *Env. Tox. and Chem.* In prep.

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