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Septic System Impacts on Groundwater Quality

William Robertson

THE
GROUNDWATER
PROJECT

Septic System Impacts on

Groundwater Quality

The Groundwater Project

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The Groundwater Project Foreword

The United Nations Water Members and Partners establish their annual theme a few years in advance. The theme for World Water Day on March 22, 2022, is "Groundwater: making the invisible visible." This is most appropriate for the debut of the first Groundwater Project (GW-Project) books in 2020, which have the goal of making groundwater visible.

The GW-Project, a non-profit organization registered in Canada in 2019, is committed to contribute to advancement in education and brings a new approach to the creation and dissemination of knowledge for understanding and problem solving. The GW-Project operates the website https://gw-project.org↗ as a global platform for the democratization of groundwater knowledge and is founded on the principle that:

"Knowledge should be free and the best knowledge should be free knowledge." Anonymous

The mission of the GW-Project is to provide accessible, engaging, high-quality, educational materials, free-of-charge online in many languages, to all who want to learn about groundwater and understand how groundwater relates to and sustains ecological systems and humanity. This is a new type of global educational endeavor in that it is based on volunteerism of professionals from different disciplines and includes academics, consultants and retirees. The GW-Project involves many hundreds of volunteers associated with more than 200 organizations from over 14 countries and six continents, with growing participation.

The GW-Project is an ongoing endeavor and will continue with hundreds of books being published online over the coming years, first in English and then in other languages, for downloading wherever the Internet is available. The GW-Project publications also include supporting materials such as videos, lectures, laboratory demonstrations, and learning tools in addition to providing, or linking to, public domain software for various groundwater applications supporting the educational process.

The GW-Project is a living entity, so subsequent editions of the books will be published from time to time. Users are invited to propose revisions.

We thank you for being part of the GW-Project community. We hope to hear from you about your experience with using the books and related materials. We welcome ideas and volunteers!

The GW-Project Steering Committee

April 2021

Foreword

The state of our groundwater resources is deteriorating due to the combination of depletion and pollution. Pollution occurs when the capacity of the subsurface to assimilate anthropogenic chemicals is exceeded and results in contaminant plumes that spread too far from the input locations. We use the groundwater environment as a waste receptor founded on the premise that the assimilation processes operating on the contaminants provide forgiveness. They provide much forgiveness but not enough to prevent deterioration of groundwater quality at many millions of locations across the globe where we leak, spill or dispose of waste in exceedance of the subsurface assimilative capacity.

One of the most striking examples of the failure of this premise comes from the widespread use of septic systems to make our household liquid wastes 'disappear'. This is the subject of this book: *Septic System Impacts on Groundwater Quality.* The idea of the septic system originated in France around 1860, shortly after Darcy discovered his famous 'law' for groundwater flow. Our reliance on septic systems achieved official engineering credibility in the 1970's via regulations of jurisdictions in the United States and beyond that standardized septic system design and installation. In the context of water treatment engineering, the purpose of the septic system is to infiltrate all of a household's waste water through the vadose zone into the groundwater zone where, according to the inherent assumption, all of the potentially harmful constituents are fully attenuated and hence the front of the septic plume does not reach groundwater beyond the household property. This was the premise in the 1950's when thousands of suburban housing developments were initiated throughout the United States, Canada and beyond.

Early on, except for nitrate and phosphate, the subsurface showed adequate assimilative capacity because typical household wastewater contained few hazardous constituents. However, the chemical makeup changed over the decades and now includes many anthropogenic constituents. In addition, the density of septic systems has increased substantially. Consequently, the formerly not so harmful concept evolved into the cause of common harm. Not only do septic plumes harm groundwater, they discharge nitrogen and phosphorus into lakes and ocean estuaries causing them to be choked with algal growth. A septic system that does not get all of the waste water into the groundwater zone is by definition a 'failed' septic system; while the other systems are deemed to be successful.

For the engineering and scientific community, the impact of the septic system on groundwater and surface waters should be a lesson in humility concerning our propensity to use the subsurface for waste disposal without rigorous prior assessment of its capability to fully attenuate harmful constituents. This book shows that small (point) sources can produce persistent and far-ranging groundwater plumes.

      The author of this book, Dr. William Robertson, Professor Emeritus, University of Waterloo, Canada, is the leading scientific authority on the impact of septic systems on groundwater from the perspective of field investigations of septic plumes. He has published widely on plume studies and has developed alternative designs for treatment of household waste waters.

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John Cherry, The Groundwater Project Leader

Guelph, Ontario, Canada, April 2021

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William Robertson

1 What is a Septic System?

In rural and many suburban areas, household wastewater is treated on-site, in disposal systems such as septic systems. Although versions of the flush toilet with associated sewer systems, have been in operation in urban centers since Roman times and earlier, in rural areas, outdoor pit privies were the common method for waste disposal until relatively recently. A privy is a small building surrounding a bench with holes through which a person may urinate or defecate. The modern 'septic system' was invented by Jean-Louis Mouras in France, around 1860. It involved indoor plumbing, with wastewater piped to a collection tank in which fats floated and solids settled while the liquid flowed to an associated drainage pit. After 10 years of operation, he was surprised to discover that the tank contained mostly liquid and remained relatively free of solids. A patent was granted in 1881, and these systems began appearing in the U.S. shortly thereafter. Their use became widespread during the 1940s, and in the 1970s, regulations were introduced in many jurisdictions, that standardized their design and installation (Van Delden, 2015). In the USA, more than 22 million septic systems are in operation, servicing ~ 25 percent of the population and it is estimated that more than 500,000,000 on-site waste-water treatment systems are in use worldwide (McCray et al., 2005; Conn et al., 2006).

Septic systems typically consist of a septic tank, which collects wastewater and an associated drainfield (also referred to as a leach field), where the wastewater is discharged into the subsurface (Figure 1).
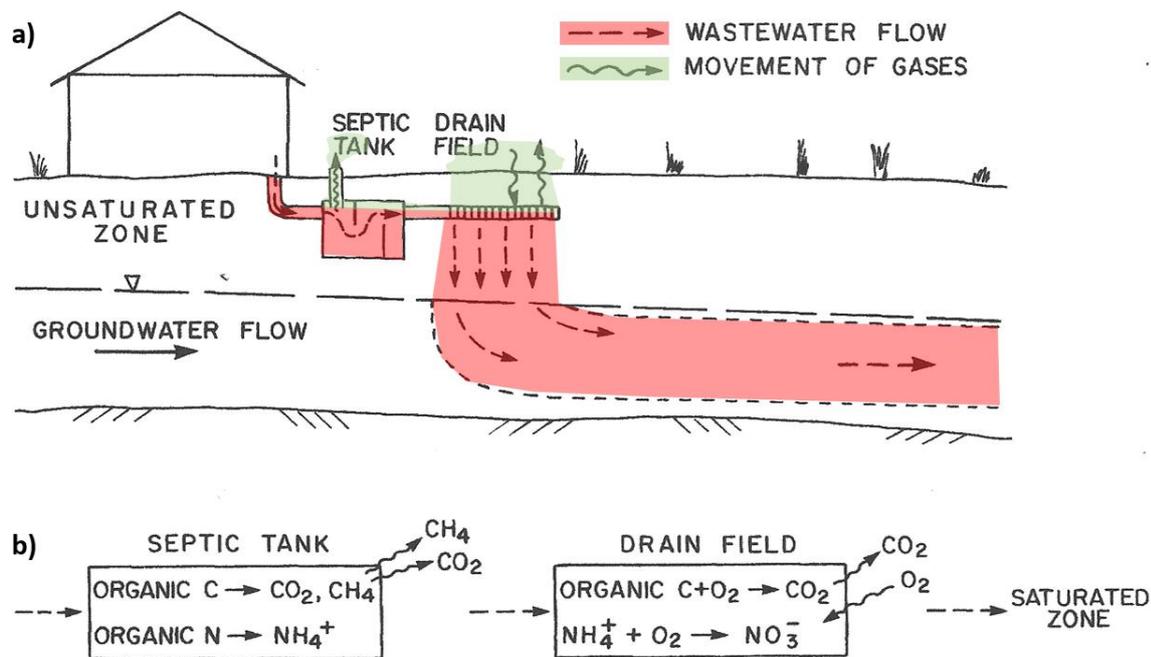

Figure 1 - Typical household septic system showing wastewater treatment steps that occur in the septic tank, and in the unsaturated sediments underlying the drainfield (adapted from Wilhelm et al., 1994).

Wastewater treatment in septic systems can be considered analogous to that provided by conventional communal treatment systems. Solids settle out in the tank and are pumped out periodically, providing the function of primary filtration, whereas the drainfield allows the filtered liquid to percolate into the subsurface where oxidation reactions occur, analogous to the oxidation provided by mechanical aeration during conventional secondary sewage treatment.

Generally, septic tanks are sized to provide several days of wastewater retention, which is the average time that the wastewater spends in the tank, so the volume of the tank is large enough to hold several days of wastewater. Drainfield size is based on the wastewater loading rate (i.e., volume of waste water per day) and the permeability of the drainfield sediments. A '*percolation test*' is a simple-to-perform, semi-quantitative estimate of permeability (OMAFRA, 2019). The percolation rate is the rate of decline of the water level in a shallow hole in the soil that will contain the drainfield. The drainfield must be large enough to allow infiltration of the daily wastewater volume each day. A household septic system located on relatively permeable sediments, typically has a septic tank volume of about 3000 liters (L) and a drainfield that is about 100 m^2 in area.

Wastewater is discharged from the septic tank by gravity flow (or by pumping) to infiltration pipes that are typically trenched into the drainfield sediments to about 0.7 m depth. This allows the wastewater to migrate through the unsaturated zone before entering the groundwater flow system at the water table (Figure 1). Typically, a drainfield consists of multiple lengths of perforated pipe installed in trenches that are backfilled with permeable pea-sized gravel. However, there are multiple drainfield designs, including importing higher-permeability sediment (usually coarse sand), at sites where low permeability sediments are present. Documents that provide more information on drainfield designs are available on-line (e.g., Septic Smart, OMAFRA, 2019). State and provincial regulations usually mandate minimum separation distances between the infiltration pipes and the water table or underlying impermeable formations. In the province of Ontario, Canada, infiltration pipes must be positioned a minimum of 1.9 m above the water table or bedrock formations (OMAFRA, 2019). Additional regulations govern the placement of drainfields, and mandate minimum separation distances: e.g., 15 m from drilled wells; 30 m from dug wells; 15m from surface water courses; and 5 m from permanent structures in Ontario (OMAFRA, 2019).

2   Potential for Contamination from Septic Tank Effluent

A compilation of septic tank effluent composition at sites treating domestic wastewater is provided in Table 1. Nitrogen in domestic wastewater occurs primarily as NH_4^+-N (nitrogen present in dissolved ammonium) and septic tank effluent values range from 18-108 mg/L. Although nitrate is generally absent in the effluent, when NH_4^+ is

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nitrified in the unsaturated zone, nitrate concentrations have the potential to exceed the drinking water limit for $NO_3^-\text{-}N$ (nitrogen present in dissolved nitrate) of 10 mg/L. The environmental impacts of groundwater with elevated levels of wastewater-derived nitrate discharging to freshwater lakes and coastal waters are an increasing concern (Persky, 1986; Harris, 1995; Cape Cod Commission, 2015). In addition, effluent phosphorus (*P*) concentrations (3-15 mg/L, Table 1), are several orders of magnitude higher than guidelines proposed to maintain surface water quality of sensitive lakes and rivers (e.g., 0.01 mg/L *P* in the USA; USEPA, 2000). Septic system effluent also exceeds drinking water criteria for pathogens and potentially a variety of other trace constituents. Consequently, considering the volume of wastewater generated by septic systems (e.g., 260 L/d/capita in the USA, McCray et al., 2005), septic systems may be considered one of the largest potential sources of groundwater contamination, worldwide. However, on-site treatment such as septic systems, provide a variety of treatment steps in the subsurface that have the potential to diminish the contaminant risk. Treatment is particularly active in the unsaturated zone beneath the drainfield. A number of the more important reactions are depicted in Figure 1 and are discussed in the following sections.

Table 1 - Septic tank effluent composition at sites treating domestic wastewater, including: electrical conductivity (EC), dissolved organic carbon (DOC), alkalinity (Alk), soluble reactive phosphorus or phosphate (SRP), the artificial sweeteners acesulfame and sucralose, and bacteria and virus colony forming units (CFU).

Parameter	Median, Mean (± std dev), or Range	Reference (n = number of samples or values)
EC (µS/cm)	1000	Robertson et al., 1991 (n=2)
	2481	Harman et al., 1996 (n = 8)
	1456 ± 314	Robertson, 2012 (n ≥ 7)
	1480 ± 131	Geary and Lucas, 2019 (n = 17)
DOC (mg/L)	46 ± 27	Robertson et al., 1998 (n = 8)
	11 ± 5.0	Withers et al., 2011 (n = 37)
	56 ± 26	Robertson et al., 2012 (n = 3)
Alk (mg/L) as $CaCO_3$	316 ± 40	Wilhelm et al., 1996 (n = 6)
	311 ± 102	Robertson et al. 1998 (n = 8)
	310 ± 105	Withers et al., 2011 (n = 37)
pH	7.1 ± 0.4	Robertson et al. 1998 (n = 8)
	7.3 ± 0.2	Withers et al., 2011 (n = 37)
	7.4 ± 0.2	Geary and Lucas, 2019 (n = 17)
$NH_4^+\text{-}N$ (mg/L)	66 ± 41	Robertson et al. 1998 (n = 9)
	4-13	USEPA, 2002
	34 ± 10	Hinkle et al., 2008 (n = 10)
	18 ± 16	Withers et al., 2011 (n = 37)
	58	McCray et al., 2005 (n = 37)
	72 ± 37	Robertson et al.,2019 (n = 111)
	108 ± 16	Geary and Lucas, 2019 (n = 14)
$NO_3^-\text{-}N$ (mg/L)	0.2 ± 0.3	Robertson et al. 1998 (n = 9)
	<1	USEPA, 2002
	0.2	McCray et al., 2005 (n = 33)
	0.03 ± 0.03	Hinkle et al., 2008 (n = 10)
	4.2 ± 3.2	Withers et al., 2011 (n = 37)
	0.2 ± 0.2	Geary and Lucas, 2019 (n = 10)

Table 1 - continued

Parameter	Median, Mean (± std dev), or Range	Reference (n = number of samples or values)
Total Nitrogen, TKN (mg/L)	26-75	USEPA, 2002
	53 ± 14	Hinkle et al., 2008 (n = 10)
Total Phosphorus (mg/L)	6-12	USEPA, 2002
	4.6 ± 4.2	Withers et al., 2011 (n = 37)
SRP (mg/L)	8.4 ± 3.5	Robertson et al. 1998 (n = 9)
	9.0	McCray et al., 2005 (n = 35)
	3.2 ± 2.6	Withers et al., 2011 (n = 37)
	8.2 ± 4.9	Robertson et al.,2019 (n = 123)
	15 ± 1.8	Geary and Lucas, 2019 (n = 16)
Cl^- (mg/L)	67 ± 64	Robertson et al. 1998 (n = 9)
	32 ± 16	Hinkle et al., 2008 (n = 10)
	54 ± 16	Withers et al., 2011 (n = 37)
	64	Robertson et al.,2019 (n = 106)
Na^+ (mg/L)	54 ± 27	Robertson et al. 1998 (n = 9)
	49 ± 29	Withers et al., 2011 (n = 37)
K^+ (mg/L)	22 ± 15	Robertson et al. 1998 (n = 9)
	26 ± 8	Withers et al., 2011 (n = 37)
Ca^{2+} (mg/L)	38 ± 36	Robertson et al. 1998 (n = 9)
	96 ± 22	Withers et al., 2011 (n = 37)
B (mg/L)	0.51 ± 0.05	LeBlanc, 1984 (n = 3)
	0.11 ± 0.03	Withers et al., 2011 (n = 37)
	0.28 ± 0.02	Bassett et al., 1995 (n = 3)
	0.49	Vengosh et al., 1994 (n = 21)
Fe (mg/L)	0.45 ± 0.41	Robertson et al. 1998 (n = 8)
	0.12 ± 0.06	Withers et al., 2011 (n = 37)
Al (mg/L)	0.1 ± 0.1	Robertson et al. 1998 (n = 6)
Acesulfame (µg/L)	57	Snider et al., 2017 (single family, n = 14)
	32	Snider et al., 2017 (communal, n = 36)
	44 ± 32	Robertson et al., 2019 (n = 56)
Sucralose (µg/L)	40 ± 25	Oppenheimer et al., 2011 (n = 8)
	51	Snider et al., 2017 (single family, n = 14)
	28	Snider et al., 2017 (communal, n = 36)
	40 ± 34	Robertson et al., 2019 (n = 56)
Fecal Bacteria (CFU/100 mL)	10^5	Viraraghavan, 1978
	10^6	Shadford et al., 1997
	10^6-10^8	USEPA, 2002
	10^5	Geary and Lucas, 2019
Coliphage Virus (CFU/100mL)	10^8	Deborde et al., 1998a

3   Fate of Pathogens During On-site Wastewater Disposal

Bacteria and viruses are present in septic tank effluent at concentrations typically in the range of 10^5 to 10^8 colony forming units (CFU)/100 mL (Table 1). These are colloid-sized particles that can be immobilized by both filtration and attachment (adsorption) during subsurface flow. The smaller size of viruses (~ 0.1 µm), compared to bacteria (~1 µm) make them less vulnerable to physical filtration, but increases their susceptibility to adsorption (Bales et al., 1995).

3.1   Bacteria

Examples of well-developed groundwater plumes below drainfields at a household site (Cambridge site) and at a campground (Long Point site) are shown in Figure 2 and Figure 3 respectively.
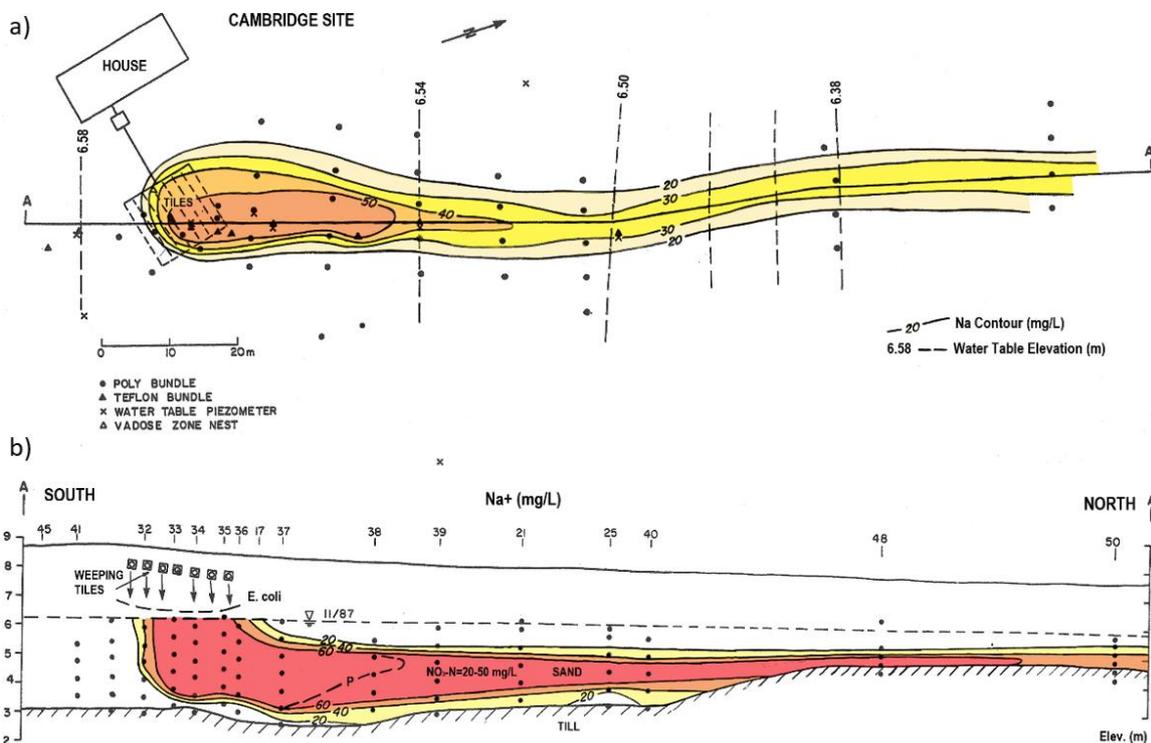
Figure 2 – A plan view (a) and cross-sectional view (b) of a groundwater plume from a household septic system at the Cambridge site in Ontario, Canada, showing: Na^+ concentration (mg/L), the extent of E. coli (> 1 cfu/100 mL) presence, and phosphate (P) zones with concentration greater than 1 mg/L, dashed lines, and range of NO_3^--N concentrations (mg/L) in the plume core. Septic tank effluent has mean Na^+ of 98 mg/L, NH_4^+-N of 30 mg/L and SRP of 8 mg/L (adapted from Robertson et al.,1991; Shadford et al., 1997; and Robertson, 2003).
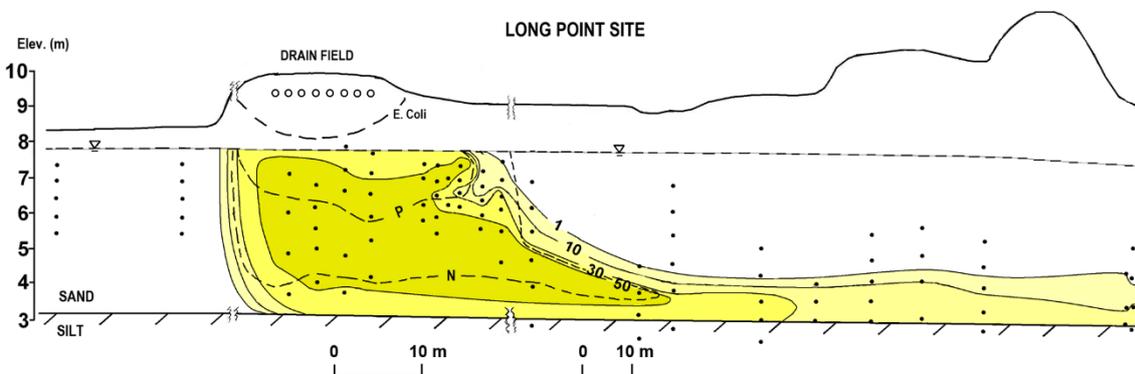
Figure 3 - Septic system plume from a large campground at the Long Point site in Ontario, Canada, showing concentrations of the artificial sweetener, acesulfame (μg/L); and the extents of the zones (dashed lines) of NO_3^--N (>1 mg/L), phosphate-P (>1 mg/L) and E coli (>10 cfu/100 mL), along the plume centerline. Dots indicate multilevel monitoring points. The septic tank effluent has mean acesulfame concentration of 50 μg/L, $NH4^+-N$ of 100 mg/L and SRP of 6.3 mg/L (adapted from Robertson et al., 2013).

Fecal bacteria are largely removed within the unsaturated zone at both sites. In a test of bacteria mobility at the Cambridge site, tracer bacteria (E. coli NAR) were dosed into the septic tank over a three-week period and then monitored in the unsaturated zone below the drainfield and in the underlying groundwater plume, over an 84-day period (Shadford et al., 1997). The tracer bacteria were detected in unsaturated zone lysimeters within 30 days of the start of the injection, but only at concentrations that were much lower than in the septic tank (~1000-fold decrease). In the groundwater plume, the tracer bacteria were almost entirely absent, with detection occurring in only one of 255 samples (Shadford et al., 1997). The Cambridge unsaturated zone is 1.5 m thick, consists of fine to medium sand, and has a wastewater retention time of about 15 days (Robertson, 1994). In similar testing at two other sites, but with shallower water tables (<1 m depth), Shadford and others detected the tracer bacteria in groundwater directly below the drainfields, but not in other monitoring wells located only a few meters farther downgradient, with the exception of a few very low concentrations. Reneau and Pettry (1975), Viraraghavan (1978), and Chen (1988) report fecal coliform concentrations in a number of household and cottage septic system plumes (at three, one and seventeen sites, respectively). However, most of these sites had shallow water tables and shallow lower permeability sediments or bedrock. Consequently, monitoring was focused in the shallow soil zone where groundwater flow likely occurred through macropores. At all of these sites, fecal coliforms were either not detected in the groundwater plumes, or were depleted by at least 2 to 3 orders of magnitude, within a few meters downgradient from the drainfields.

In contrast, evidence for rapid transport of bacteria has been shown at sites with more favorable hydrologic conditions. In a tracer test conducted in the Cape Cod municipal wastewater plume, where relatively rapid groundwater velocities of 0.2-0.7 m/day occur in the medium to coarse sand aquifer, only a 10-fold reduction of a bacterial tracer occurred over a travel distance of 6 m and the arrival of the bacterial tracer coincided with the arrival of the *Br* co-tracer (Bales et al., 1995).

3.2   Viruses

Viruses are an order of magnitude smaller in size compared to bacteria, (~0.1 vs 1 µm) and consequently, are less impeded by physical filtration during subsurface flow. However, they remain strongly affected by sorption reactions as a result of their net negative surface charge at normal pH ranges (Bales et al., 1995). Virus sorption (attachment) is influenced by factors such as sediment particle size, organic carbon content, water content, solution pH and ionic strength. Attachment is considered the most important depletion process for viruses because it leads to inactivation (die-off) and is irreversible in some cases (Betancourt et al., 2019).

In large-scale laboratory sand columns, Betancourt and others (2019) observed variable transport of fecal viruses, with die-offs ranging from 3000-fold over 4 meters to

1000-fold over 0.3 m, at rates that were non-linear (first order) with rate constants ranging from 200 to 1000 day^{-1}. Such rapid depletion implies removal by attachment (sorption).

Deborde and others (1998a), reported a high degree of fecal coliphage virus attenuation in the groundwater plume from a school septic system at a site that had rapid groundwater velocities. The unsaturated zone was 1.5 m thick and the permeable sand and gravel aquifer had groundwater velocities of 1 to 3 m/day. Coliphage virus decreased 100,000-fold, within 7 m of the drainfield and further declined to below detection (< 1 CFU/100 mL) beyond 38 m from the drainfield. However, in a subsequent test in which both tracer viruses and bromide were injected into the core of the plume, a portion of the tracer virus arrived at monitoring locations 7 to 17 m downgradient, at about the same time as Br$^-$, after only 2 to 6 days (Deborde et al., 1998b). However, concentrations were 100,000 times lower over this distance, similar to the die-off observed in the ambient plume. It was also determined that the virus retained in the aquifer sediments near the injection well, remained viable for more than nine months after injection. In another test in the same aquifer, a tracer virus was injected at a location 22 m away from a large-scale pumping well and a large fraction of the injected virus mass (up to 55%) was captured by the well within 47 hours (Deborde et al., 1999; Woessner et al., 2001), illustrating the potential for rapid viral transport under certain conditions.

Rapid viral transport was also observed in the Cape Cod tracer experiment discussed above. During that experiment, a bacteriophage virus co-tracer arrived at the monitoring location 6 m downgradient from the injection site, about 12 days after the injection, coincident with Br$^-$ and the tracer bacteria. However, the tracer virus was more highly attenuated compared to the tracer bacteria (100,000-fold versus 10-fold depletion) because of its greater susceptibility to removal by adsorption. However, during a second phase of the same tracer test, when higher pH water was injected, a substantial amount of the adsorbed virus mass was remobilized (Bales et al., 1995).

Overall, field observations indicate that pathogens, including both bacteria and viruses are relatively immobile in septic system plumes. However, under certain conditions they can be more mobile, particularly when fast groundwater velocities occur due to the presence of high permeability sand and gravels, fractures or macropores; or to close proximity of high-yield production wells.

4   Septic Effluent Chemistry and Treatment

Properly designed septic systems treat wastewater prior to reaching the water table. Wastewater treatment occurs in the septic tank and in the drainfield, as water percolates through the unsaturated zone.

4.1   Treatment in the Septic Tank (Anaerobic)

Septic systems usually collect all household wastewater in a single tank that is partitioned into two sections (Figure 1), so that solids settle out in the first compartment. Domestic wastewater is a rich source of labile organic carbon and consequently, the effluent normally becomes anaerobic during storage in the septic tank. Anaerobic conditions can lead to a number of reactions in the tank, including organic nitrogen mineralization to NH_4^+ as shown in Equation 1 (Wilhelm et al., 1994).

$$CO(NH_3^+)_2 \,(urea) + H_2O \rightarrow 2NH_4^+ + CO_2 \tag{1}$$
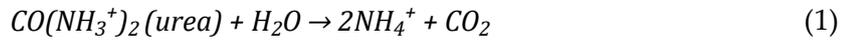
Anaerobic conditions also lead to ammonium, NH_4^+, conversion to NH_3 gas as shown in Equation 2 (Wilhelm et al., 1994).

$$NH_4^+ + OH^- \rightarrow NH_3 + H_2O \tag{2}$$

In addition, anaerobic conditions result in sulfate reduction as shown in Equation 3 and methane generation as shown in Equation 4 (Wilhelm et al., 1994).

$$SO_4^{2-} + 2CH_2O + 2H^+ \rightarrow H_2S + 2CO_2 + 2H_2O \tag{3}$$

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O \tag{4}$$
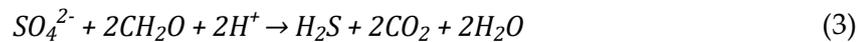
Gases generated as a result of these reactions (CO_2, NH_3, H_2S, CH_4) are usually removed using a ventilation stack as shown in Figure 1, which is repeated here for the reader's convenience. These gases are assumed to rapidly oxidize in the atmosphere. In some cases, the drainfield is not vented, and gases are conveyed to the leach field, where they migrate upward from the perforated drainfield piping.
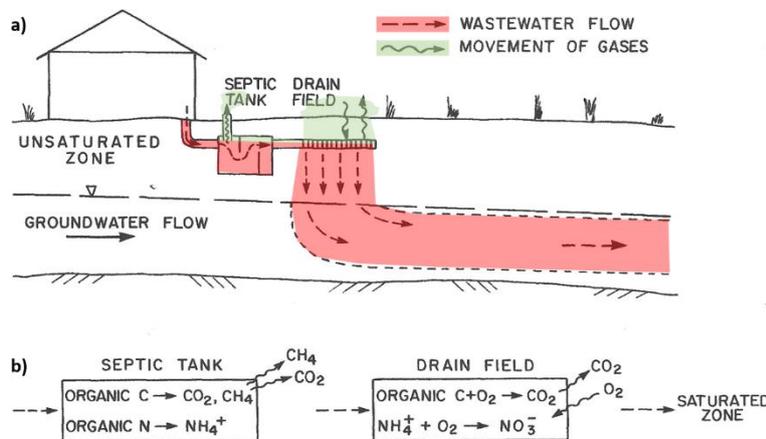
Repeat of Figure 1 - Typical household septic system showing wastewater treatment steps that occur in the septic tank, and in the unsaturated sediments underlying the drainfield (adapted from Wilhelm et al., 1994). This repeat of Figure 1, provides an opportunity to notice the gases that are released into the atmosphere from the vent pipe or through the soil above the drainfield.

4.2   Treatment in the Unsaturated Zone (Aerobic)

Normally, drainfields are designed to discharge the effluent into the unsaturated zone where oxidation of the wastewater can occur. Important oxidation reactions include conversion of organic carbon to CO_2 (Equation 5 and 6) and nitrification of NH_4^+ as shown in Equation 7 (Wilhelm et al., 1994).

$$CH_2O + O_2 \rightarrow CO_2 + H_2O \tag{5}$$

$$CO_2 + H_2O \rightarrow H_2CO_3 \rightarrow H^+ + HCO_3^- \tag{6}$$

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \tag{7}$$
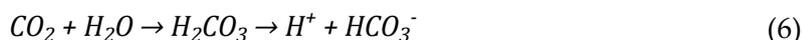
Table 2 compares dissolved organic carbon (DOC) concentrations in septic tank effluent and the associated groundwater plumes at a number of sites. DOC removal in the drainfields at these sites, ranges from 22 to 97 percent. Greater removal occurs at sites where the wastewater is well oxidized, as indicated by the relatively complete conversion of NH_4^+ to NO_3^-.

Table 2 - Comparison of dissolved organic carbon (DOC) concentrations in septic tank effluent and associated groundwater plumes, illustrating organic carbon degradation in the unsaturated zone below the drainfields. Plumes are listed in order of increasing levels of wastewater oxidation as indicated by decreasing NH_4^+ concentrations in the plumes. Plume values are mean (or representative) values from the proximal plume zones underlying the drainfields, unless indicated otherwise.

Site	Effluent DOC (mg/L)	Groundwater Plume DOC (mg/L)	DOC removal (%)	NH_4^+-N (mg/L)	NO_3^--N (mg/L)	Reference
JL, cottage	83	44	47	73	0.1	Robertson et al., 1998
LP2, campground	56	11	80	48	50-80	Robertson et al., 2012
LP2, campground	56[1]	6.3 (5-8)	89			Aukes et al., 2019
SM, campground	~50[2]	13	~75	37	0.1	Garda, 2018
Otis, municipal wastewater	19	7[3]	63	14	0.1	LeBlanc, 1984
DEL resort, reducing	9	7	22	3	0.1	Robertson et al., 1998
LP1, campground	21	5 ± 0.8 (n=8)	76	1	51	Robertson & Cherry, 1992
PP, campground	32	13	78	0.4	18	Robertson et al., 1998
DEL resort, oxidizing	9	5	45	0.2	13	Robertson et al., 1998
LA, school	28	0.7	97	0.1	131	Robertson et al., 1998
CA, house	40	4	90	0.1	14	Robertson et al., 1998
MU, house	81	3	96	0.1	37	Robertson et al., 1998

[1]Effluent DOC value is from Robertson et al., 2012.
[2]Effluent DOC value is not available for the SM campground site, but is assumed to be similar to the value at the LP2 campground site, because the two sites have similar effluent Cl$^-$ and NH_4^+ concentrations.
[3]Plume values are from a monitoring well located 1000 m from the drainfield.

The oxidation reactions generate acidity, but this is usually buffered through a series of rock-water interactions. In calcareous terrain, the dissolution of carbonate minerals

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(Equation 8), normally restores near- neutral pH conditions and in most cases, also results in increased Ca^{2+} concentrations in the plume.

$$CaCO_3 + H^+ \rightarrow Ca^{2+} + HCO_3^- \tag{8}$$

In granitic terrain, in the absence of carbonate minerals, buffering can be provided by *Al-* and *Fe*-oxyhydroxide minerals (e.g., gibbsite and ferrihydrite) that are usually present as minor constituents in most sediments. However, these minerals are relatively insoluble at neutral pH values. Their dissolution, as shown in Equations 9 and 10, becomes important only under low pH conditions.

$$Al(OH)_3 + 3H^+ \rightarrow Al^{3+} + 3H_2O \tag{9}$$

$$Fe(OH)_3 + 3H^+ \rightarrow Fe^{3+} + 3H_2O \tag{10}$$

Septic system plumes in granitic terrain, when well oxidized, are often acidic, with pH values commonly in the range of 4 to 6.

5   Septic System Plume Formation and Dispersion

When the infiltrating wastewater encounters the water table, it enters the local groundwater flow system, forming a plume that is distinct from the background groundwater. Examples of two well-characterized plumes from septic systems in unconfined sand aquifers of Ontario, Canada are shown in Figures 2 and 3. Figure 2b and Figure 3 are repeated here for the readers convenience.
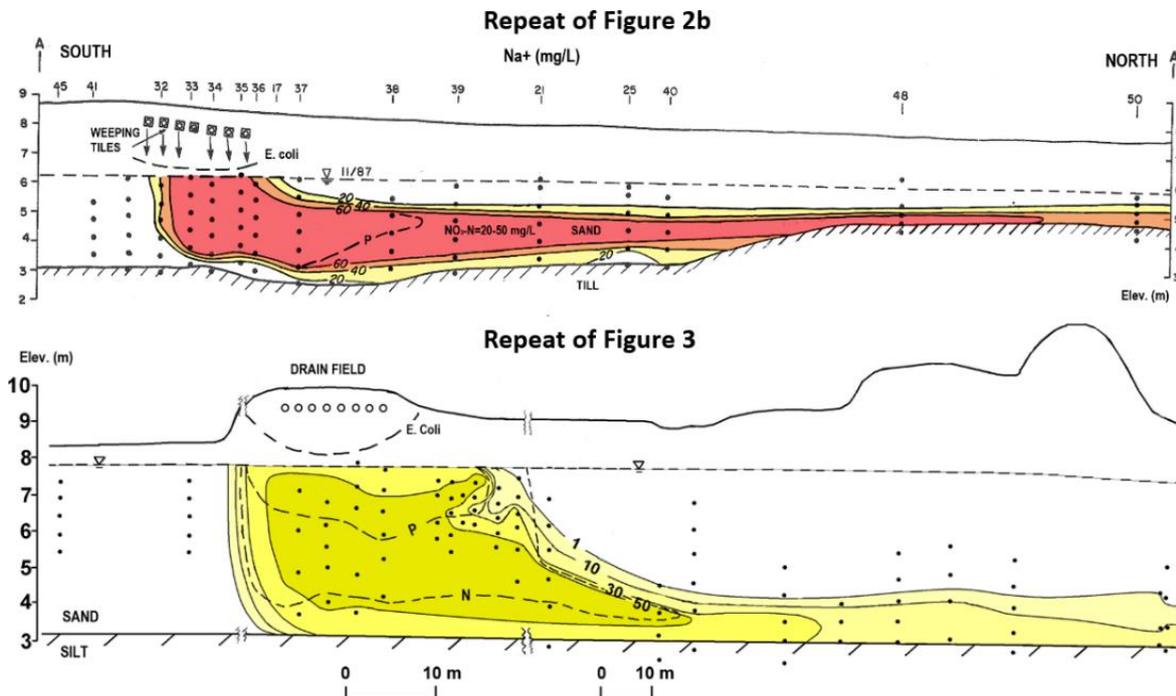
Repeat of Figure 2b and Figure 3 – Repeated for the reader's convenience. Figure 2b shows a household septic system plume in Cambridge, Ontario, Canada, and was adapted from Robertson and others (1991). Figure 3 shows a campground septic system plume in Long Point, Ontario, Canada, and was adapted from Robertson and others (2013).

Figure 2 shows a plume from a household septic system that was mapped using Na^+ as the plume tracer. Plume boundaries remain sharp throughout the 200 m length of the plume that was mapped, and Na^+ concentrations remain essentially undiluted in the core of the plume. Model sensitivity analyses as shown in Figure 4, indicate that the persistence of sharp plume boundaries, with an absence of core zone dilution, requires low values for both horizontal and vertical transverse dispersivity (< 1 cm). The modeled values are consistent with low dispersivity values determined from natural-gradient tracer tests in other sand aquifers (Sudicky et al., 1986; Moltyaner and Killey, 1988; Garabedian et al., 1991).
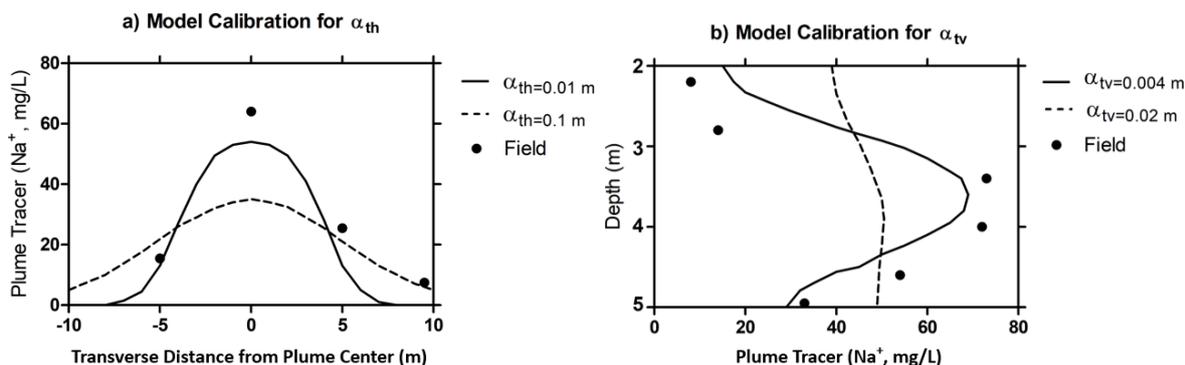

Figure 4 - Cambridge septic system plume showing model calibrations for: a) horizontal transverse dispersivity and b) vertical transverse dispersivity. Dots indicate Na^+ concentrations measured: a) in a monitoring fence located 95 m from the drainfield and aligned transverse to the direction of groundwater flow and b) in a multilevel sampling bundle located 30 m from the drainfield. Dashed and solid lines indicate Na^+ concentrations predicted using a three-dimensional analytical advection-dispersion model with differing values for horizontal (α_{th}) and vertical (α_{tv}) transverse dispersivity (from Robertson et al., 1991).

The second plume example as was shown in Figure 3 is from a septic system at a large campground (Long Point site) and shows a groundwater plume that again, has sharp boundaries, and an absence of core dilution throughout the 200 m mapped length of the plume. In this case, the plume was traced using an artificial sweetener (acesulfame) that is currently used in a wide range of calorie-reduced food and beverage products. Concentrations throughout the plume core are > 18 μg/L, which is about 10,000 times higher than the background detection limit value at this site (< 0.01 μg/L). Such an unusually high contrast between background and effluent concentrations allows very small inputs from septic system wastewater to be detected as discussed further next.

An important factor to consider when monitoring septic system behavior is the thickness of the septic system plume. Plume thickness is governed by the wastewater loading rate, drainfield size and the ambient average linear groundwater velocity. It can be estimated using the relationships shown in Equation 11 and Equation 12.

$$b = (q * t)/porosity \tag{11}$$

$$t = w/vel \tag{12}$$

where:

b = plume thickness at the downgradient edge of the drainfield (L)

q = wastewater loading rate per unit area of the drainfield (L/T)

$porosity$ = saturated aquifer porosity

t = groundwater residence time under the drainfield (T)

w = width of the drainfield in the direction of groundwater flow (L)

vel = ambient average linear groundwater velocity (L/T)

For example, a typical household septic system receiving 1 m^3/day of wastewater discharged to a drainfield that is 10 m by 10 m in plan view, would have a wastewater loading rate of 0.01 m/day. For an average linear groundwater velocity of 0.2 m/day, groundwater would reside under the drainfield for 50 days, because a drop of water moving at 0.2 m/day would require 50 days to traverse the 10 m length of the drainfield. For an aquifer porosity of 0.33, the plume would be 1.5 m thick (ignoring dispersion). This requires careful vertical placement of sampling locations for effective monitoring. Both the Cambridge and Long Point sites shown in Figures 2 and 3, are located close to groundwater flow divides, where groundwater velocities are relatively low (~0.07 to 0.08 m/day, Robertson et al., 2019). These low velocities, combined with a higher wastewater loading rate at Long Point (0.06 m/day, Robertson et al., 2019), have resulted in the formation of plumes that are relatively thick (~3 to 5 m), compared to plumes at many other sites.

The low capacity for dispersion in the Cambridge and Long Point aquifers, as well as in many other aquifers (e.g., Weiskel and Howes, 1991), allows relatively high concentrations of wastewater constituents to persist in the plume core zones. Thus, unless attenuation reactions occur, relatively long distances of travel may be required before contaminants are reduced to acceptable levels by dilution with background groundwater.

5.1   Plume Tracers--Major Ions; (EC, Cl⁻, NO₃⁻, Na⁺)

Septic system plumes usually have elevated concentrations of major ions such as Cl^-, Na^+, NO_3^- and sometimes NH_4^+, derived from the wastewater, as well as increased alkalinity (Equation 6) and other constituents resulting from rock-water interactions in the drainfield (e.g., Ca^{2+}, Equation 8). All of these constituents are potentially useful as plume tracers. Figure 2 shows that sodium values in the plume core at the Cambridge site (>50 mg/L) are about five times higher than background values (< 10 mg/L). Furthermore, elevated concentrations of dissolved constituents impart increased specific conductance (EC) to the plume water and as a result, septic system plumes can usually be identified in the field using a specific conductance meter. Table 1 provides a summary of typical major ion concentrations occurring in septic tank effluent at sites where domestic wastewater is

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being treated. The main limitation for using major ions as plume tracers is that all of these constituents are not unique to domestic wastewater. Fertilizers used in modern agricultural operations contain potash and nitrogen formulations that can lead to elevated K^+, Cl^- and NO_3^- levels in impacted groundwater, of the same magnitude as those found in septic system plumes (Spalding and Exner, 1993; Goss et al., 1998). In northern climates, de-icing salts applied to roadways and parking lots in winter, can lead to Cl^- and Na^+ concentrations similar to those found in septic system plumes (Baer et al., 2019). In the Cambridge plume (Figure 2) for example, neither Cl^- nor NO_3^- could be used directly as plume tracers, because similar concentrations occurred in the background groundwater, as a result of fertilizer use on the adjacent farm fields. In the Long Point plume (Figure 3), Cl^- use as a tracer was limited because background values were elevated as a result of the use of $CaCl_2$ on the campground roadways to suppress dust. In both the Cambridge and Long Point plumes, however, Na^+ and EC remained effective as plume tracers.

Whereas major ions may serve as useful plume tracers in the proximal plume zones, in more distal zones, their usefulness is often diminished because of the presence of similar contaminants from other sources. This circumstance can limit the usefulness of major ions as indicators of septic system impact in situations where impacts from multiple contaminant sources may overlap, such as may occur in water withdrawn by wells from broad contributing areas (e.g., Figure 5).
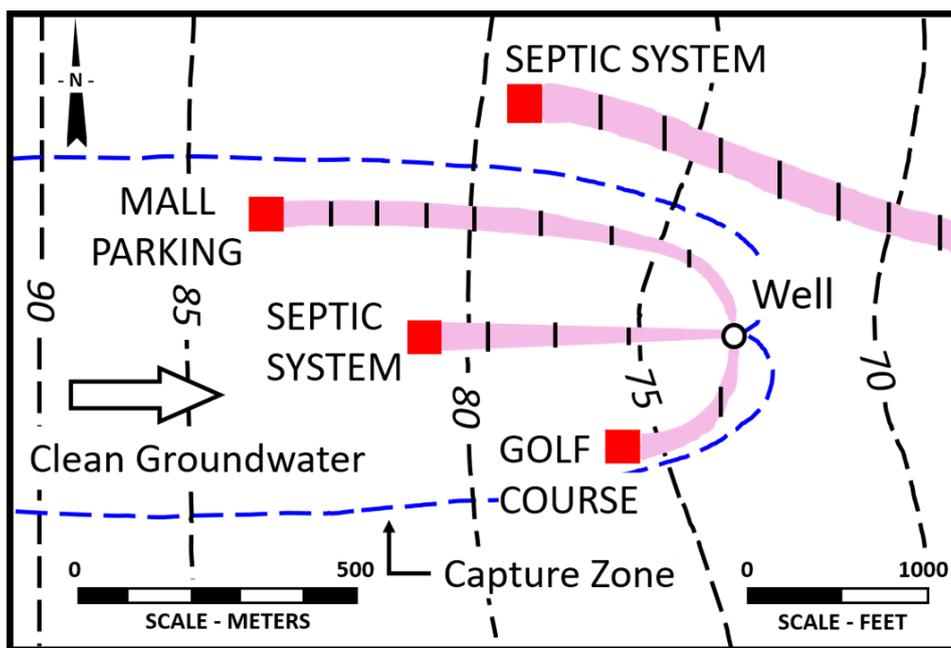

Figure 5 - Conceptual view of the capture zone of a water supply well encountering a septic system plume and other potential sources of similar contamination, including a shopping mall parking lot that contributes sodium and chloride from de-icing salts, and a golf course that contributes nitrate from fertilizer. Adapted from Einerson and Mackay (2001).

5.2   Plume Tracers--Minor and Trace Constituents--Boron

Boron (*B*) is present in laundry detergents, is mobile in groundwater flow systems, and has been used as a tracer to identify septic system and municipal wastewater impacts in groundwater (LeBlanc, 1984; Vengosh et al., 1994; Bassett et al., 1995). However, Boron concentrations in domestic wastewater (ranging from 0.1 to 1 mg/L, Table 1) are elevated by a relatively small factor (e.g., about 5, Bassett et al., 1995) compared to background values, and some natural waters, such as brackish and irrigation-impacted groundwater, can have *B* values equally as high as wastewater. Consequently, boron is only modestly robust as a wastewater tracer, but is useful at some sites. In a municipal wastewater plume on Cape Cod, Massachusetts, *B* was used as one of the principal tracers because it occurred throughout the 3 km length of the plume, at consistent concentrations of 0.2-0.4 mg/L, which was about five times higher than background values (LeBlanc et al., 1984; Barbaro et al., 2013).

5.3   Plume Tracers--Artificial Sweeteners (Acesulfame and Sucralose)

The artificial sweetener potassium acesulfame ($C_4H_4KNO_4S$) is used in a wide range of calorie-reduced food and beverage products and is now ubiquitous in most domestic wastewaters throughout the world (Buerge et al., 2009). This compound is relatively unique to domestic wastewater and occurs in septic tank effluent at concentrations typically in the range of 10-100 µg/L (Table 1). Acesulfame is relatively persistent during treatment in both municipal wastewater plants (Buerge et al., 2009) and in septic systems (Van Stempvoort et al., 2011a), as well as in surface water and groundwater flow systems (Buerge et al., 2009; Spoelstra et al., 2013; Spoelstra et al., 2017). Considering its typically high concentration in wastewater, compared to much lower background values in groundwater (e.g., ~10,000 times higher in the Long Point plume, Figure 3), it offers the potential to be a powerful tracer of septic system impact. Sucralose ($C_{12}H_{19}Cl_3O_8$) is another widely used artificial sweetener that also resists degradation, and occurs in domestic wastewater at concentrations similar to acesulfame (Table 1). It also has potential for use as a tracer (Oppenheimer et al., 2011). In the Long Point plume shown in Figure 3, acesulfame was used as a principal tracer because other potential tracers such as Cl^- were elevated in the background groundwater at this site, as a result of the use $CaCl_2$ as a dust suppressant on the campground roadways. Interference from other sources is a common problem associated with wastewater tracers (e.g., Figure 5) and highlights the usefulness of artificial sweeteners because they occur primarily only in domestic wastewater.

The use of specific artificial sweeteners has also changed over the past several decades and the presence or absence of specific sweeteners in groundwater plumes can serve as a rudimentary age dating tool. Potassium acesulfame has been in widespread use only since 1988 and sucralose since 2000. In a study of contaminant plumes and leachate at

15 landfill sites, Roy and others (2014) found that acesulfame was present in high concentrations and was the dominant sweetener at seven of eight active sites, but at seven sites that were decommissioned prior to 1990, saccharin ($C_7H_5NO_3S$) was the dominant sweetener and acesulfame was detected only at low concentrations.

More recently studies have revealed, that in some cases, acesulfame can be degraded during wastewater treatment (Castronovo et al., 2017; Kahl et al., 2018). This raises the possibility that acesulfame may also be subject to degradation in subsurface environments. Although a number of studies have demonstrated its persistence at relatively high concentrations, these studies do not preclude the possibility of slow degradation in certain environments, and further studies should be undertaken.

5.4   Isotopic Tracers--Nitrate-^{15}N

Several constituents in septic system plumes (e.g., Cl^-, B, NO_3^-) can be isotopically distinct from other sources, therefore isotopic characterization can assist in identifying groundwater impacted by septic systems. Nitrate-^{15}N ($NO_3^- \text{-}^{15}N$) values have been utilized to distinguish between groundwater nitrate contamination resulting from septic systems and contamination from fertilizers used in agricultural operations. Nitrate derived from chemical fertilizers has lower ^{15}N values, typically in the range of +3 to +5 per mil (parts per thousand), whereas NO_3^- in septic system plumes generally has higher ^{15}N values in the range of +7 to +14 per mil (Kreitler et al., 1979; Heaton, 1986; Aravena et al., 1993; Wassenaar, 1995). However, there is some overlap in the isotopic composition of these sources and processes such as NH_4^+ volatilization, soil zone exchange, and denitrification, can alter the isotopic signature of the source material. In addition, when manure is used in agricultural operations, the isotopic signature of the associated NH_4^+ can be similar that of domestic wastewater. At some sites it has been noted that groundwater $NO_3^- \text{-}^{15}N$ values tend to cluster near the range where fertilizer and wastewater $NO_3^- \text{-}^{15}N$ values overlap (+5 to +8 per mil) making source identification using isotopic signatures difficult (Komor and Anderson, 1993; Xue et al., 2009). However, the lighter isotopes of NO_3^- (^{14}N and ^{16}O) are preferentially consumed during nitrate degradation processes such as denitrification and anammox as discussed in Section 6, and consequently, isotopic analyses can be of considerable value in establishing when such processes are active.

6   Fate of Nitrogen in Septic System Plumes

At sites where the wastewater has been well oxidized, septic system plumes often have nitrate concentrations that exceed the drinking water limit of 10 mg/L as $NO_3^- \text{-}N$. For example, in the Cambridge plume, effluent $NH_4^+\text{-}N$ averaging 30 mg/L, is completely oxidized in the unsaturated zone and as a result, there is an absence of NH_4^+ in the plume,

but NO_3^--N is elevated, and ranges from of 16-38 mg N per liter (Figure 2). Figure 6 shows a histogram of nitrate concentrations measured in 21 septic system plumes in Ontario, and shows that the drinking water limit was exceeded in 16 of the plumes.
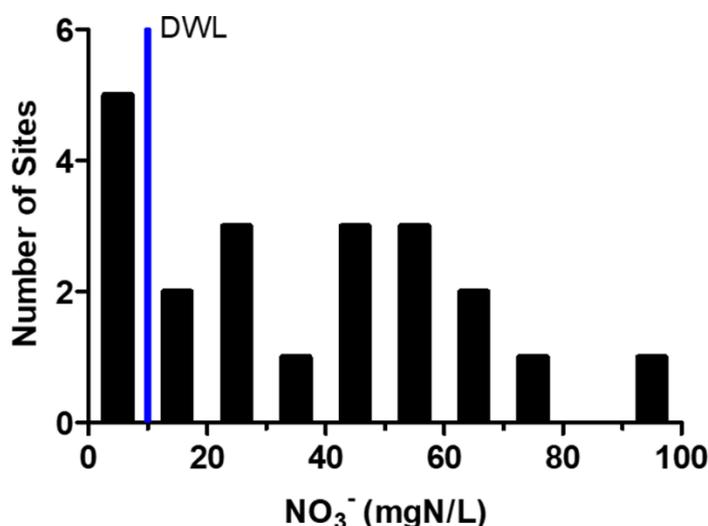

Figure 6 - Histogram of mean NO_3^- concentrations in proximal plume zones underlying drainfields at 21 Ontario septic system sites (from Robertson et al., 2020). The nitrate drinking water limit (DWL) of 10 mg N per liter is shown for reference.

Groundwater in densely developed coastal areas of Cape Cod, Massachusetts, USA has NO_3^--N concentrations as high as 24 mg/L, and this is largely attributed to the widespread use of septic systems (Coleman et al., 2018).

Nitrate is potentially reactive however, and can be reduced to di-nitrogen gas (N_2) if reducing conditions are encountered and if potential electron donor compounds are available to act as energy sources to support the transformation reactions. Two commonly occurring NO_3^- transformation reactions are heterotrophic denitrification using labile organic carbon as the energy source (Equation 13) and autotrophic denitrification using reduced sulfur compounds (e.g., pyrite) as the energy source (Equation 14).

$$4NO_3^- + 5CH_2O \rightarrow 2N_2 + 4HCO_3^- + CO_2 + 3H_2O \tag{13}$$

$$14NO_3^- + 5FeS_2 + 4H^+ \rightarrow 5Fe^{2+} + 10SO_4^{2-} + 7N_2 + 2H_2O \tag{14}$$

These reactions tend to occur abruptly at reaction fronts, rather than uniformly along the plume (Smith et al., 1991). The reaction fronts usually occur at locations where the plumes encounter increased concentrations of electron donor compounds, such as may occur in riparian zone sediments enriched in organic carbon (e.g., Robertson et al., 1991; Böhlke et al., 2002) or in deeper aquifer zones where unweathered pyrite may be present (e.g., Postma et al., 1991; Kölle et al., 1985). The Cambridge plume shown in Figure 2, is suboxic (i.e., dissolved oxygen < 1 mg/L) and could potentially be affected by nitrate transformation reactions. However, NO_3^- concentrations remain consistently elevated and show no

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evidence of isotopic enrichment, indicating that denitrification is inactive (Aravena et al., 1993). Presumably, this is because there is an absence of suitable electron donor compounds (e.g., labile organic carbon or unweathered pyrite) in the aquifer sediments at this site.

In contrast, at the Long Point site (Figure 3), the proximal plume zone has elevated NO_3^--N of 32 to 79 mg/L (Aravena and Robertson, 1998) and nitrate is completely consumed to < 1 mg/L within 40 m downgradient from the drainfield. Transformation occurs along a reaction front where there are coinciding increases in both alkalinity and SO_4^{2-}, indicating that heterotrophic denitrification (Equation 13), as well as autotrophic denitrification (Equation 14), contribute to the observed nitrate loss. At this site, the denitrification is promoted by the presence of trace amounts of labile organic carbon and pyrite in the deeper (unweathered) aquifer sediments (Aravena and Robertson, 1998). The Long Point aquifer is a recent beach deposit that is less than a few thousand years old, whereas the Cambridge aquifer was deposited at the end of the last glacial period, ~10 to 15 thousand years before present. The younger age of the Long Point aquifer, as well as the presence of wetland complexes nearby, may contribute to the richer reserve of labile electron donor compounds at that site. Autotrophic denitrification is known to occur in aquifers and aquitards worldwide because trace amounts of pyrite (or other reduced sulfur compounds) are usually present when the sediments are unweathered (Kölle et al., 1985; Postma et al., 1991; Korom, 1992; Robertson et al., 1996). In most cases this reaction is marked by distinct increases in SO_4^{2-} concentrations of groundwater in the same zone where NO_3^- values decline.

A third potentially important nitrate transformation reaction that has been recognized recently, is NO_3^- reduction by NH_4^+ (anammox, Equations 15 and 16).

$$NO_2^- + NH_4^+ \rightarrow N_2 + 2H_2O \tag{15}$$

$$3NO_3^- + 5NH_4^+ \rightarrow 4N_2 + 9H_2O + 2H^+ \tag{16}$$

These reactions can be active at sites where the wastewater is less well oxidized, such that both NO_3^- and NH_4^+ occur together in the plume. The reaction is characterized by isotopic enrichment of both the residual NO_3^- and NH_4^+ (Clarke et al., 2008; Caschetto et al., 2017) and by the presence of anammox-specific bacteria (Moore et al., 2011; Smith et al., 2015). Isotopic and bacteriological characterization has shown that anammox is an important contributor to nitrate transformations observed in a number of septic system plumes, including the Long Point plume (Robertson et al., 2012), the Killarney site campground plume (Caschetto et al., 2017) and the large municipal wastewater plume on Cape Cod (Smith et al., 2015). In the Cape Cod plume, anammox bacteria were found to be present throughout a 3 km-long zone where NH_4^+ was present, and tracer injection tests established that NO_3^- consumption from anammox and denitrification were of a similar magnitude and occurred at similar rates, when labile DOC was available. However, when DOC supply was

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limited, NO_3^- consumption became dominated by anammox, accounting for about 90% of the observed NO_3^- loss. Furthermore, in a review of total inorganic nitrogen (TIN) persistence in the 21 Ontario septic system plumes (Robertson et al., 2020), it was discovered that TIN consumption was significantly greater at 10 sites where both NO_3^- and NH_4^+ were present in the plume zones near the drainfields (62 percent TIN loss), compared to other sites where only NO_3^- was present (3 percent TIN loss). Overall, this evidence suggests that the anammox reaction occurs widely in septic system plumes, and it rivals denitrification in importance for providing consumption of nitrate in these plumes.

The fourth potential nitrate transformation reaction is dissimilatory nitrate reduction to ammonium (Equation 17), is considered less important in groundwater environments (Rivett, et al., 2008) and has not been reported in septic system plumes.

$$NO_3^- + 2CH_2O + 2H^+ \rightarrow NH_4^+ + 2CO_2 + H_2O \tag{17}$$
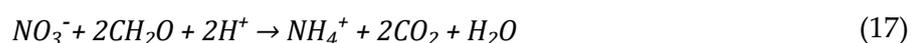
7   Fate of Phosphorus in Septic System Plumes

The fate of phosphorus in septic system plumes is dependent on acidity of water in the unsaturated zone and reactions involving phosphorus minerals. The first two, short, subsections of this section provide the most common chemical reactions that are relevant to acidity of unsaturated zone water and precipitation of minerals in the unsaturated zone that contribute to removal of phosphorus from the water. Section 7.1 describes how acid is buffered in the unsaturated zone and Section 7.2 describes common precipitation reactions involving phosphorus. The fate of phosphorus is discussed in Section 7.3.

7.1   Acid Buffering Reactions

Four common acid buffering reactions are important to controlling pH in the unsaturated zone which in turn influences the fate of phosphorus in septic system plumes. These acid buffering reactions involve dissolution of carbonate minerals and gibbsite, as well as two types of ferrihydrite dissolution. These reactions are as follows:

- carbonate mineral dissolution as shown in Equation 18;

$$CaCO_3 + H^+ \rightarrow HCO_3^- + Ca^{2+} \tag{18}$$

- gibbsite dissolution as shown in Equation 19;

$$Al(OH)_3 + 3H^+ \rightarrow Al^{3+} + 3H_2O \tag{19}$$

- ferrihydrite dissolution as shown in Equation 20;

$$Fe(OH)_3 + 3H^+ \rightarrow Fe^{3+} + 3H_2O \tag{20}$$

- and, reductive dissolution of ferrihydrite as shown in Equation 21.

$$4Fe(OH)_3 + CH_2O + 7H^+ \rightarrow 4Fe^{2+} + HCO_3^- + 10H_2O \tag{21}$$
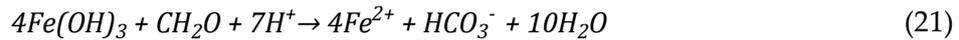
Lower pH increases the potential for reactions presented in Section 7.2 that precipitate phosphorus, thus remove phosphorus from the waste water. Consequently, at sites where the reactions shown in Equations 18 through 21 are more likely to occur, there is a greater assimilation capacity for phosphorus in the unsaturated zone below a septic system.

7.2   Reactions involving Phosphorus Minerals

Four reactions involving phosphorus minerals are relevant to the fate of phosphorus in septic system plumes, including precipitation of hydroxyapatite, variscite, strengite and vivianite. These reactions are as follows:

- hydroxyapatite precipitation as shown in Equation 22;

$$5Ca^{2+} + 3HPO_4^{2-} + H_2O \rightarrow Ca_5(PO_4)_3OH + 4H^+ \tag{22}$$

- variscite precipitation as shown in Equation 23;

$$Al^{3+} + PO_4^{3-} + 2H_2O \rightarrow AlPO_4 \cdot 2H_2O \tag{23}$$

- strengite precipitation as shown in Equation 24;

$$Fe^{3+} + PO_4^{3-} + 2H_2O \rightarrow FePO_4 \cdot 2H_2O \tag{24}$$

- and, vivianite precipitation as shown in Equation 25;
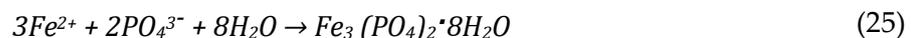
$$3Fe^{2+} + 2PO_4^{3-} + 8H_2O \rightarrow Fe_3(PO_4)_2 \cdot 8H_2O \tag{25}$$

There is a greater assimilation capacity for phosphorus in the unsaturated zone below a septic system at locations where conditions promote the precipitation reactions presented in Equations 22 through 25.

7.3   Phosphorus Fate

Phosphorus concentrations in septic tank effluent (e.g., 5 to 15 mg/L total *P*, Table 1) are several orders of magnitude higher than values capable of stimulating algal growth and eutrophication in aquatic environments (~0.03 mg/L, Dillon and Rigler, 1974). Total phosphorus in septic tank effluent consists of a combination of dissolved phosphate (PO_4^{3-}), often referred to as soluble reactive phosphorus (SRP), organic phosphorus that occurs as a component of the organic compounds present in the wastewater, and phosphorus adsorbed onto particulate material in the effluent. In septic tank effluent, SRP usually comprises 70-85% of total P (McCray et al., 2005). During infiltration through the drainfield

sediments, particulate material is removed by filtration and organic compounds are degraded, and as a result, total *P* in septic system plumes is mostly SRP (Harman et al., 1996). SRP concentrations in septic system plumes vary, but some sites have *P*-rich zones where SRP concentrations approach values found in the septic tank effluent, including zones below the drainfields at both the Cambridge and Long Point sites, where SRP values range from 1 to 6 mg/L (Figures 2 and 3). One of the largest reported phosphorus plumes occurs in the Cape Cod municipal wastewater plume, where a *P*-rich zone with SRP values of 1 to 3 mg/L, extends 600 m downgradient from the wastewater infiltration beds, before discharging to a small lake (LeBlanc et al., 1984; McCobb et al., 2003).

Phosphate is a trivalent anion that can be influenced by a number of reactions that make its fate in the subsurface complex. In most groundwater flow systems, phosphorus is strongly affected by surface-layer sorption reactions because of the presence of minerals with net positive surface charges (e.g., calcite, gibbsite, ferrihydrite at normal pH ranges) in most sediment types. Sorption has the effect of slowing the rate of *P* migration, but does not permanently immobilize *P*. The potential for sediment to adsorb phosphorus is typically characterized by a distribution coefficient (K_d), which describes the ratio of the mass of *P* adsorbed on the sediment solids, to the mass in solution. K_d values for phosphorus can be determined from laboratory testing of sediment samples and are typically in the range of 3 to 40 cm^3/g (Walter et al., 1996; McCray et al., 2005). The retardation equation (Equation 26, from Freeze and Cherry, 1979), relates K_d values to a retardation factor for *P* migration in groundwater.

$$R = 1 + \left(\frac{\rho_b}{\theta}\right) K_d \tag{26}$$

where:

R = phosphorus retardation factor (average linear groundwater velocity)/(phosphorus migration velocity)

ρ_b = sediment dry bulk density (g/cm^3)

θ = porosity (-)

K_d = distribution coefficient; *P* mass adsorbed/*P* mass in solution (cm^3/g)

The range of K_d values noted above, lead to *P* retardation factors in the range of 15 to 400 for aquifers with typical sediment properties (e.g., sediment density of 2.65 g/cm^3 and porosity of 0.3). In a review of *P* mobility in 24 Ontario septic system plumes (including the 21 sites noted above), seven sites with well-developed *P* plumes had *P* retardation factors that ranged from 11 to 67 (Robertson et. al., 2019), which is generally consistent with the range expected based on K_d values reported in the literature.

In addition to surface-layer sorption, chemical equilibrium models predict that mineral precipitation reactions involving phosphorus should also be active in septic system plumes (Ptacek, 1998; Parkhurst et al., 2003; Spiteri et al., 2007). The above 24 site review

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(Robertson et al., 2019) observed that distinct zones of sediment *P* accumulation were present in almost all of the drainfields, and these *P*-rich zones consistently occurred within 1 to 2 m below the infiltration pipes, even in systems that varied widely in age. Consistent accumulation near the infiltration pipes, suggested that this was the result of mineral precipitation reactions, rather than sorption. Furthermore, sand grains from the *P* accumulation zones exhibited distinct secondary mineral coatings and these coatings contained phosphorus (Figure 7).
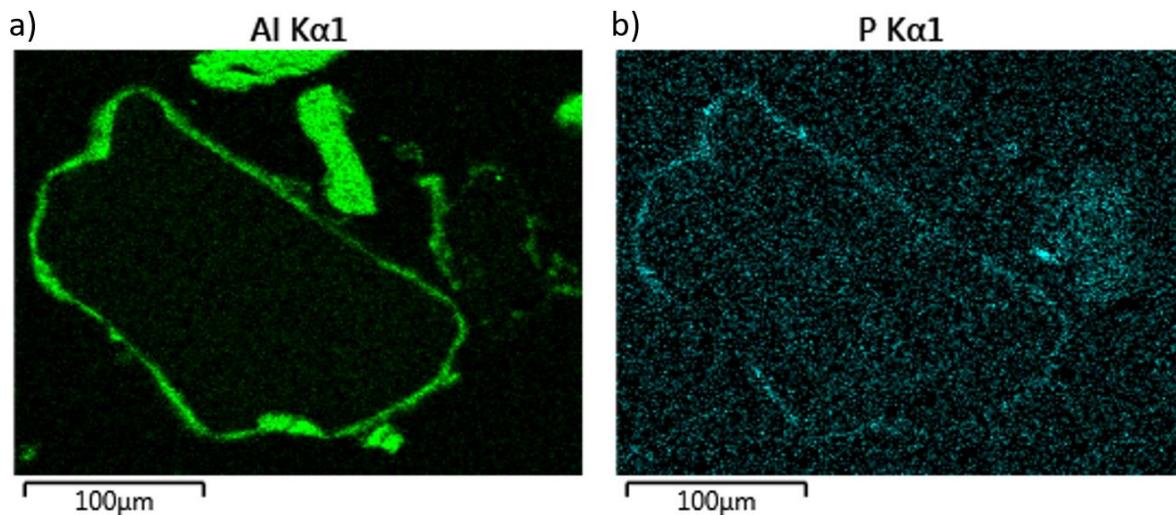

Figure 7 - Secondary *P*-mineral coating on a drainfield quartz grain from the filter bed at a campground in Ontario, Canada: a) scanning electron microscope image and b) x-ray elemental analysis (Robertson et al., 2019). The coating composition is: *Al* 19.4, *Fe* 6.6, *P* 2.0 and *Ca* 1.8 percent by weight. (Image courtesy of Surface Science Western, London, Ontario, Canada).

The composition of the coatings was variable, but iron, aluminum and occasionally calcium, were the dominant cations, indicating a relationship to the precipitation reactions noted above (Equations 22 to 25).

The acidity generated by wastewater oxidation, particularly the oxidation of NH_4^+ (Equation 7), can have an important effect on the fate of phosphorus. Decreases in pH can lead to increased dissolution of gibbsite (Equation 19) and ferrihydrite (Equation 20) which can lead to increased concentrations of Al^{3+} and Fe^{3+}. This, in turn, can stimulate the precipitation of variscite (Equation 23) and strengite (Equation 24). This relationship is supported by drainfield porewater analyses at several sites, including the Cambridge site, where zones of sediment *P* accumulation coincide closely with the zones where wastewater NH_4^+ is being nitrified (Figure 8).
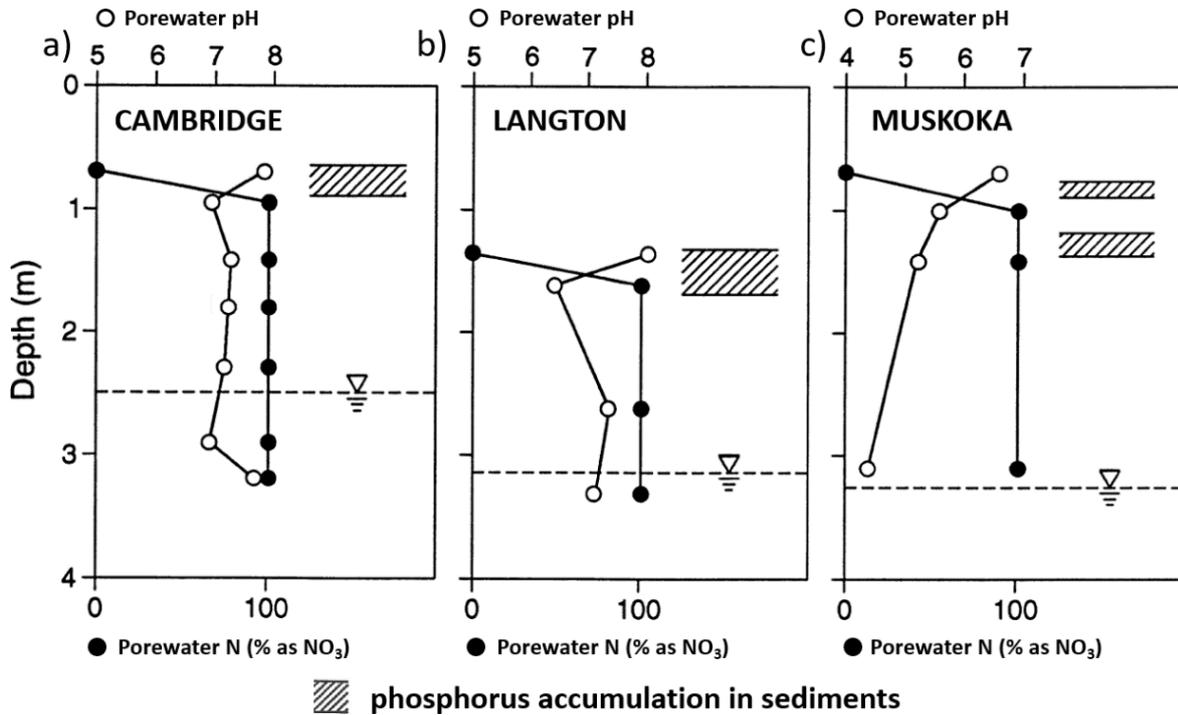
Figure 8 - Porewater pH and nitrate prevalence in drainfield sediments at three septic system sites in Ontario, Canada where sand aquifers are present: a) Cambridge and b) Muskoka both household systems, and c) Langton public school septic system. Porewater NO_3^--N values indicate percent of total inorganic nitrogen ($NH_4^+ + NO_3^-$-N) occurring as NO_3^-. Samples with no NO_3^- are the septic tank effluent (100 percent NH_4^+-N) and indicate the depth position of the drainfield infiltration pipes. Nitrate and pH values from the unsaturated zone are from porewater squeezed from undisturbed sediment cores (from Zanini et al., 1998). Also shown, as hash-marked depth intervals, are the zones where phosphorus has accumulated in the drainfield sediments.

In granitic terrain, acidic conditions can persist in septic system plumes and pH values are commonly between 4 and 6. In the review study noted above (Robertson et al., 2019), *P* removal was found to be much greater at the sites on granitic terrain where acidic plumes were present (90 percent *P* removal), and plume SRP values showed a deceasing trend with lower plume pH values (Figure 9).
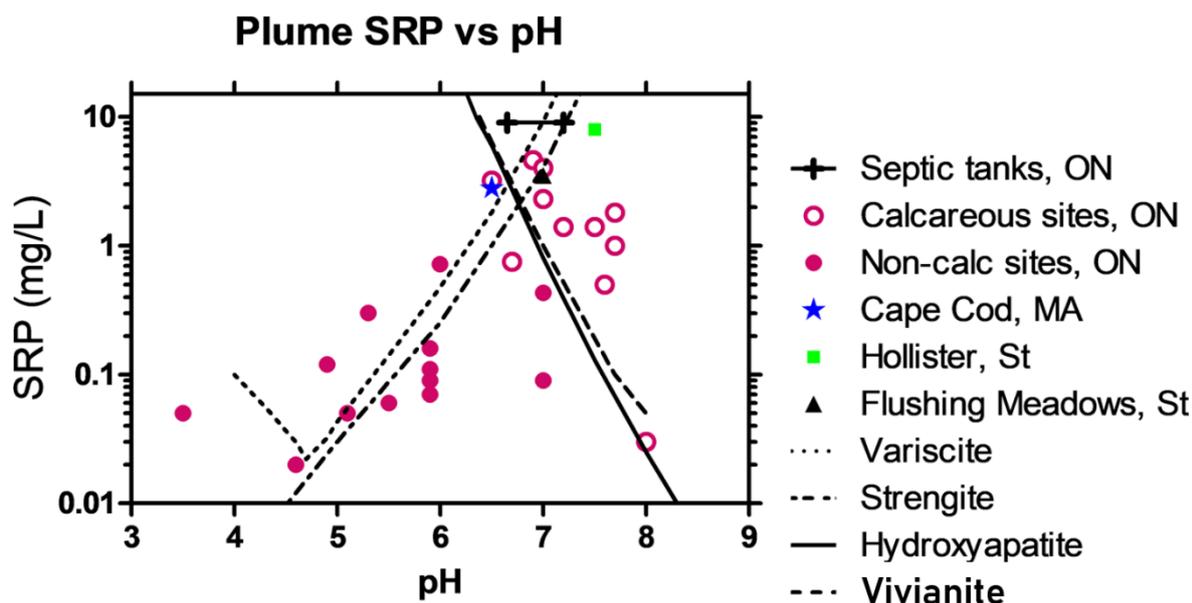
Figure 9 - Mean or representative soluble reactive phosphorus (SRP) concentrations in the proximal zones of 27 septic system plumes with differing pH values, compared to values predicted considering the solubility of the *P*-minerals hydroxyapatite, variscite, strengite and vivianite; Open and filled dots (24 Ontario, Canada sites) from Robertson et al. (2019); Cape Cod, USA site from Bussey and Walter (1996); Hollister and Flushing Meadows, USA sites from Robertson (1995). Solubility curves were determined using the chemical equilibrium model, PHREEQC (Parkhurst, 1995) and assume equilibrium with gibbsite and ferrihydrite, *Ca* concentration of 90 mg/L and for the vivianite solubility curve, ferrous *Fe* concentration of 1 mg/L (adapted from Robertson et al., 1998). Also shown as the horizontal black line ("Septic tanks, ON" in the legend) is the median septic tank effluent SRP value (8.4 mg/L) and range of effluent pH values at 22 of the Ontario sites (from Robertson et al., 2019).

Phosphorus accumulation in the drainfield sediments has also been observed at other locations, including the Cape Cod municipal wastewater infiltration site (Walter et al., 1996) and at sites in Sweden (Eveborn et al., 2014). The relatively high concentrations of phosphorus that occur in septic tank effluent (i.e., 5 to 15 mg/L, Table 1), compared to most other aqueous environments, increases the likelihood that mineral precipitation reactions will occur.

8 Fate of Trace Organic constituents in Septic System Plumes

A number of trace organic constituents can be present in septic tank effluent at concentrations that are of environmental and health concern, and consequently, their treatment in septic systems is of interest. Effluent from the Cambridge site shown in Figure 2, for example, was found to contain more than 30 volatile organic compounds (VOCs), mostly substituted benzenes, related to oil and grease residues introduced during dish washing, clothes laundering, and other cleaning tasks. However, none of these compounds were detected (i.e., none were present at a concentration greater than 1 µg/L) in the underlying groundwater plume, indicating relatively complete VOC transformation in the 2-m thick, sandy, unsaturated zone at this site (Robertson, 1994). In a tracer injection test conducted in the same study, dichlorobenzene contained in a plumbing line cleaner, was

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dosed into the septic tank, and was also found to be almost entirely degraded in the unsaturated zone. The drainfield unsaturated zone at the Cambridge site, provides a high degree of DOC removal (3 mg/L in the plume versus 38 mg/L in the effluent, Table 2) and provides complete oxidation of the wastewater NH_4^+ (Robertson, 1994). Swartz and others (2006) measured a suite of pharmaceutical compounds and estrogenic substances in the plume from a residential septic system on Cape Cod, Massachusetts, USA and found that several pharmaceutical compounds, including caffeine and paraxanthine, were highly consumed in the 3-m thick, sandy, unsaturated zone at that site, but that several estrogenic substances, including 17β-estradiol and estrone, persisted in the groundwater plume at concentrations that were close to the septic tank effluent values.

Carrara and others (2007) measured a suite of 12 pharmaceutical and trace organic constituents in the plumes from three large septic systems in Ontario, Canada including the Long Point site shown in Figure 3, and found persistence of several compounds, including ibuprofen, gemfibrozil and naproxen. The widely prescribed anti-convulsive drug, carbamazepine, is also known to persist during sewage treatment, and in subsurface wastewater plumes (Clara et al., 2004; Heberer et al., 2004; Nakada et al., 2008), and consequently, has also been proposed as a potential wastewater tracer. However, this compound is not ubiquitously present in all domestic wastewater sources (Oppenheimer et al., 2011), which reduces its robustness as a tracer. Some other tracer candidates, such as the artificial sweeteners, acesulfame and sucralose, are more ubiquitously present.

Overall, properly functioning septic systems provide a high degree of removal of trace organic constituents, particularly in the drainfield unsaturated zones, although some recalcitrant compounds can persist. However, these same compounds also persist during conventional sewage treatment as well.

9   Failing Septic Systems

The previous sections have outlined the relatively high degree of treatment that properly functioning septic system can provide for many wastewater constituents. Nonetheless, studies have documented impacts on surface water courses at sites where septic systems are in use (Iverson et al., 2018; Withers et al. 2011; Arnscheidt et al., 2007). In most cases, data suggested that the stream impacts were the result of seepage of untreated wastewater from 'failing' septic systems. Septic system failure is the term used when wastewater discharged to a drainfield does not percolate into the subsurface, but rather, breaks out onto the surface and drains into a nearby surface water course. This can result from inadequate permeability of the drainfield sediments, from high water table conditions, or from mechanical failures that allow direct hydraulic connection to surface drainage ditches or drain pipes. For example, Withers and others (2011) documented soluble *N* and *P* concentrations that were 3 to 12 times higher in a headwater stream in the United Kingdom, at a location immediately downstream from a village where septic

systems were in use. The site was on clay rich soils where high water table conditions were present and elevated nutrient values were observed in drainage ditches and pipes that serviced the village. Furthermore, dissolved N occurred primarily as NH_4^+, rather than as NO_3^-, as would be the case for untreated wastewater leaking from a failing septic system.

10 Exercises

Exercise 1

You are thinking about purchasing a home in a rural area that has water supplied from a well that is screened over an interval 5 to 7 m below the water table. Upgradient from the property, there is a roadside motel/restaurant complex that is serviced by a septic system. You are concerned about potential for contamination of the well, so you have accessed the building permit records, and determined that the septic system drainfield is 30 m x 30 m in area with the downgradient edge located 300 m from the well. The drainfield is designed for wastewater loading of 1 cm/day (365 cm/yr). The site is underlain by a regional unconfined sand aquifer that has an average linear groundwater velocity of 100 m/yr. Precipitation, in this area, averages 90 cm/yr. Water balance calculations for a local river basin have established that evapotranspiration averages 60 cm/yr. Assuming that the aquifer has a saturated porosity of 0.3, and has sufficiently high transmissivity that pumping from the well does not materially alter flow paths, should I be concerned about the potential for contamination from the upgradient motel/ restaurant septic system?

<div align="right">Click for solution to exercise 1⤵</div>

Exercise 2

The Long Point septic system plume (Figure 2) has NO_3^--N of about 50 mg/L, but this nitrate is completely attenuated at a reaction front located about 20 m downgradient of the drainfield. The aquifer contains trace amounts of reduced S (0.02 percent by weight) reflecting the presence of biogenic pyrite (Aravena and Robertson, 1998), and this material contributes to autotrophic denitrification (Equation 14).

The average linear groundwater velocity in this plume zone has been estimated to be 28 m/yr (Robertson, 2008). Assuming that all of the NO_3^- loss is attributable to autotrophic denitrification utilizing the reduced sulfur compounds, how fast will the nitrate plume advance, based on the rate of S consumption? The aquifer has a dry bulk density of 1.8 g/cm³ and saturated porosity of 0.3.

<div align="right">Click for solution to exercise 2⤵</div>

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Exercise Solutions

Solution Exercise 1

Answer: yes

Explanation:

Drainfield loading:

$$q = wastewater\ loading + net\ recharge\ (i.e., precipitation - evapotranspiration)$$

$$q = 3.65\frac{m}{yr} + \left(0.9\frac{m}{yr} - 0.6\frac{m}{yr}\right) = 3.95\frac{m}{yr}$$

Groundwater will reside under the drainfield for:

$$residence\ time\ under\ drainfield = \frac{width\ field}{average\ linear\ velocity} = \frac{30\ m}{100\frac{m}{yr}} = 0.3\ yr$$

Plume thickness at the downgradient edge of the drainfield will be:

$$b = \frac{q * t}{porosity}$$

$$b = \frac{3.95\frac{m}{yr}0.3\ yr}{0.3} = 3.95\ m\ \tilde{}\ 4\ m$$

Downgradient from the drainfield the average linear vertical velocity will derive from precipitation recharge:

$$average\ linear\ vertical\ velocity = \frac{net\ recharge\ rate}{porosity} = \frac{0.9\frac{m}{yr} - 0.6\frac{m}{yr}}{0.3} = 1\frac{m}{yr}$$

Given the average linear lateral velocity of 100 m/yr, the septic system plume will take 3 years to reach the well.

$$time\ to\ reach\ well = \frac{300m}{100\frac{m}{yr}} = 3\ yr$$

Ignoring dispersion, during those three years, the plume would be driven vertically downward by the net recharge. With the average linear vertical velocity of 1 m/yr, the 4-m thick plume from the septic field would occur at a depth of 3 to 7 m below the water table at the well location, and could thus pass through the well screen that spans from 5 to 7 m below the water table.

Solution Exercise 2

Assume one-dimensional flow for the purpose of calculations and consider the rate of S consumption in a 1 L cube of aquifer material (10 cm x 10 cm x 10 cm).

Determine the amount of S in the 1 L aquifer cube which weighs:

$$weight\ of\ 1\ L\ cube = 10cm\ 10cm\ 10cm\ 1.8\frac{g}{cm^3} = 1800\ g$$

We know S is 0.02 percent of the dry material by weight, so:

$$grams\ of\ S\ in\ 1\ L\ cube = 1800\ g\ \ 0.0002 = 0.36\ g\ S\ in\ 1\ L\ cube$$

NO_3^--N flux through the cube is determined as follows.

$$Darcy\ Velocity = (Interstital\ Groundwater\ Velocity)\ Porosity = 2800\frac{cm}{yr}\ 0.3 = 840\frac{cm}{yr}$$

$$Volumetric\ Flow\ through\ cube = (Darcy\ Velocity)\ (Flow\ Area)$$

$$= 840\frac{cm}{yr}\ 10cm\ 10cm = 84000\frac{cm^3}{yr}$$

NO_3^--N flux through cube $= (Volumetric\ Flow)\ (Concentration)$

$$= 84000\frac{cm^3}{yr}\ \frac{50\ mg\ NO_3^-\text{-}N}{1000\ cm^3} = 4200\frac{mg\ NO_3^-\text{-}N}{yr} = 4.2\frac{g\ \ NO_3^-\text{-}N}{yr}$$

$$\frac{Moles\ \ NO_3^-\text{-}N}{yr} = 4.2\frac{g}{yr}\ \frac{1\ M\ \ NO_3^-\text{-}N}{14\ g} = 0.3\frac{M\ \ NO_3^-\text{-}N}{yr}$$

From Equation 14, 1 M of NO_3^--N will consume 10/14 M S:

$$Moles\ S\ consumed\ = 0.3\frac{M\ NO_3^-\text{-}N}{yr}\ \frac{10\ M\ S}{14\ M\ NO_3^-\text{-}N} = 0.21\frac{M\ S}{yr}$$

$$\frac{g\ S\ consumed}{yr} = 0.21\frac{M\ S}{yr}\ \frac{32.06g\ S}{1\ M\ S} = 6.87\frac{g\ S}{yr}$$

Given that there is 0.36 g S in each 1 L cube of aquifer, each cube has sufficient S to support autotrophic denitrification of a plume for 0.052 years as shown below.

$$Number\ of\ years\ required\ to\ deplete\ S\ from\ cube\ = \frac{0.36\ g\ S}{6.9\frac{g\ S}{yr}} = 0.052\ yr$$

Thus, the NO_3^--N plume will advance 10 cm every 0.052 yr, or one could say it can consume all S from 19.2 cubes each year (1/0.052=19.2), which is a rate of NO_3^--N plume advance of 192 cm/yr. Another way to calculate the advance rate is as follows.

$$NO_3^-\text{-}N\ plume\ advance\ rate\ = \frac{10\ cm}{0.052\ yr} = \frac{192\ cm}{yr}$$

[Return to Exercise 2]

About the Author
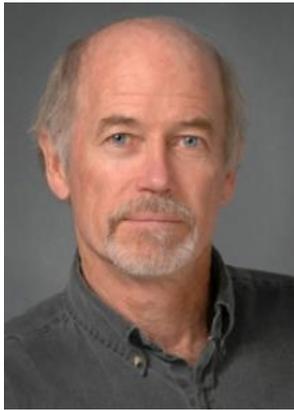
Dr. Will D. Robertson is Professor Emeritus in the Department of Earth and Environmental Sciences, University of Waterloo in Ontario, Canada, where he taught courses on physical hydrogeology and field methods in hydrogeology over a 20-year period. During that time, he advised more than 30 graduate and undergraduate students and published over 50 journal manuscripts related to groundwater age dating techniques and contamination issues associated with mine tailings, on-site wastewater disposal, and agriculture. He is also co-inventor of several patents related to enhancing nitrogen and phosphorus removal in agricultural runoff and in septic systems.

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Modifications from Original Release

page 3, Table 1, 7th row from the bottom 108 + 16 was changed to 108 ± 16.

page 19, last equation is 25 not 24 (there were two 24s and no 25) .

page 19, last sentence "Equations 22 through 24" should be "Equations 22 through 25"