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Contribution of phosphorus to Georgian Bay from groundwater of a coastal beach town with decommissioned septic systems

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ABSTRACT

Groundwater inputs of phosphorus (P) to the Laurentian Great Lakes are poorly known, but may contribute to eutrophication and algal bloom issues. This study's objective was to assess the contribution of P to Nottawasaga Bay from the surficial sand aquifer at Wasaga Beach, representing a coastal cottage area with decommissioned septic systems, and how this might change with time. The first part of the study involved site-scale groundwater sampling beside 4 provincial park public washrooms. Legacy P plumes were detected at two of these sites, with one being >30 years since decommissioning. P transport calculations including sorption onto aquifer sediments indicate the majority of P plumes from the town's decommissioned septic systems have likely not yet reached the shoreline, >50 years since installation, and will likely contribute P to the bay for many decades. The second part of the study consisted of broader-scale (town-wide) surveys of shallow beach groundwater. Dissolved P concentrations were ~50 μg/L for background groundwater (in town and reference area), which is similar to literature values. This P may have been sourced from degrading organic matter, bird droppings, or soil-aquifer minerals. Sporadic elevated concentrations up to 420 μg/L may be from legacy septic systems and/or natural sources. A rough calculation suggests groundwater P loading along Nottawasaga Bay's eastern shore (Wasaga Beach, 10-km; adjacent similar beaches, 40-km) is a few percent at most of that from the Nottawasaga River. Thus, it more likely affects localized periphyton and macrophyte growth rather than significantly affecting the Nottawasaga Bay P budget.

Introduction

It has long been recognized that domestic wastewater treatment systems (i.e., septic systems) can supply nutrients, including phosphorus (P), to nearby lakes (Dillon and Rigler 1975; Dillon et al. 1993). Phosphorus is generally perceived as the nutrient most limiting to primary production in lakes, with excess inputs leading to eutrophication and algal blooms (Schindler et al. 1971; Correll 1998; Schindler et al. 2016). This was especially a concern for small freshwater lakes of the Canadian Shield, the catchments of which typically have shallow soils on bedrock. As a result, in the 1970s, the Lakeshore Capacity Model (LCM) was developed to predict the ice-free total phosphorus (TP) concentration in such lakes, considering natural and anthropogenic (largely human wastewater from septic systems) inputs (see review by Paterson et al. 2006). In the subsequent decades it has been modified to account for new knowledge on P sources and cycling processes and to accommodate changes in human activities (e.g., per capita water use, P removal from detergents). This empirical model does not distinguish different pathways for septic P transport to the lake (i.e., relatively slow groundwater flow through overburden sediments versus more rapid flow through or over fractured bedrock) and may not be applicable in other geologic areas (Paterson et al. 2006), such as those with thick unconsolidated-sediment aquifers in southern Ontario. Furthermore, other inputs of P with groundwater are ignored (Paterson et al. 2006), as has been common in many lake P balance studies in the past (Lewandowski et al. 2015).

Many studies have reported on groundwater plumes of P derived from septic systems (e.g., Rea and Upchurch 1980; Robertson et al. 1991; Harman et al. 1996; Ptacek 1998; Robertson et al. 1998; Roy et al. 2009), and wastewater lagoons (McCobb et al. 2003) in permeable sediment aquifers. In a review, Robertson (2003) explained how mineralogy of soil-aquifer materials controls whether P in leachate reaches the water table, with acidic conditions generated in non-calcareous materials promoting phosphorus mineral precipitation in the oxidized unsaturated zone below the septic infiltration bed. In contrast, septic system effluents do not become acidic in calcareous aquifer materials, due to their acid neutralizing ability. These systems tend to leach a substantial portion of their P to the water table and, thus, produce P plumes in groundwater. Dissolved P that reaches the groundwater zone is known to sorb to aquifer materials (e.g., Robertson 1995; Harman et al. 1996; Robertson 2008), retarding the P plume compared to other

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wastewater components such as chloride. Sorption may fit a linear equilibrium model, being rapid and reversible (Robertson 2008). Other processes may present as less reversible or irreversible sorption-like reactions, including intra-particle diffusion, incorporation into the mineral phase, or precipitation of phosphate minerals, though these were not apparent in the P plume study (Long Point site; 16 years post-installation) of Robertson (2008).

In urbanizing areas, individual septic systems are often decommissioned in favor of community wastewater treatment facilities. Decommissioning involves pumping out the septic tank, disconnecting from the dwelling, and connecting to the communal sewer network. Robertson and Harman (1999) reported on 2 septic system plumes on calcareous sand aquifers and noted essentially unchanged concentrations of dissolved phosphorus in groundwater 2–4 years after decommissioning. For one plume, all other major dissolved plume constituents (e.g., Na, Ca, Cl, NO_3) returned to background levels within one year (these were not measured for the other plume). The maintenance of dissolved phosphorus concentrations was attributed to rapid and reversible sorption reactions in the groundwater zone retarding P transport, and suggests potentially long-term persistence of the P plumes.

Phosphorus has been targeted as a key driver of algal blooms in the Laurentian (Canada-U.S.) Great Lakes (International Joint Commission 2014). In general, groundwater contributions of nutrients to the Great

Fig. 1. Maps showing the location of the town of Wasaga Beach a) in the Great Lakes Basin and b) along the shore of Nottawasaga Bay; along with c) its current footprint, showing urban areas (shaded grey) and the years when certain areas had sewers installed and cottage septic systems decommissioned; also shown are the comfort station sites (profiling groundwater transects) and areas of groundwater sampling with multi-level wells (WA1–10) and shoreline surveys (WA1–12).

Lakes and their tributaries is not well understood (Robinson 2015). Georgian Bay, part of Lake Huron, contains one of the world's longest freshwater beaches, which is lined with cottages and residences that currently have septic systems or had them in the past. Beach sands in this area are reported to be calcareous, suggesting that septic-sourced P will likely be mobile, reaching the water table and travelling with groundwater. The town of Wasaga Beach, on Nottawasaga Bay of Georgian Bay in Ontario (Fig. 1), Canada, formally relied exclusively on septic systems for wastewater treatment. The majority of the community (population of 17,537; Canada Statistics 2012) has resided within 1 km of the bay along a 10-km length of shoreline (of the 14-km long freshwater beach). However, community sewer services were implemented in conjunction with septic system decommissioning beginning in 1980 at the northeast end of the beach, progressing westward and south in several stages over the next several decades (Fig. 1c), until all septic systems along the beach strip were finally decommissioned in 2007. Currently there is no information on P inputs via groundwater from the cottage areas of the freshwater sand beaches of Nottawasaga Bay.

Study Objective and approach

The objective of this study was to assess the potential contribution of P to Nottawasaga Bay from groundwater of the surficial sand aquifer at Wasaga Beach, representing a coastal cottage area with decommissioned septic systems, and how this might be changing with time. This included determining whether septic systems form P plumes in this beach sand aquifer, and how these P plumes respond to decommissioning. Further, potential septic P mass loadings to the lake were compared to other groundwater P sources. If these legacy septic systems represent a significant source of P loading to Nottawasaga Bay via groundwater, this impact should presumably first become reduced at the east end of the beach where these systems have been decommissioned longer and thus more of their P plumes may have already been flushed from the aquifer. The findings should be applicable to the other sand beaches along Nottawasaga Bay (Fig. 1b), as well as many other freshwater beaches of similar composition around the world. The findings should also provide guidance on how decommissioned septic systems should be accommodated in lake P budget models, such as the LCM.

The study consisted of 2 parts. The first part involved site-scale groundwater sampling, focusing on the septic systems of 4 provincial park comfort stations (i.e., public washrooms). These had their septic beds decommissioned along the same temporal gradient as the rest of the town; the 4 sites (CS-2A, -3A, -5, -6; Fig. 1c) span ~7 km of beach and up to 20 years difference in decommissioning times (Table 1). The identification of any groundwater P plumes at these sites would provide information on i) whether P from septic systems can reach the water table and form a groundwater plume in these beach sands, and ii) how long P plumes of decommissioned septic systems may persist.

Table 1
Details of the four comfort station sites and any existing P plume at each. Plume velocity estimates for CS-2A and -3A assume plumes existed and have been flushed beyond the sample transect.

Site	Years since sewer connection	Transect distance from septic source (m)	Detected P plume?	Plume velocity estimate (m/a)
CS-2A	30–35 (1979–1984)	20 or more	N	>0.6
CS-3A	30–35 (1979–1984)	40 or more	N	>1.1
CS-5	34 (~1980)	35–45	Y	<1.3
CS-6	14–18 (1996–2000)	15–25	Y	<1.8

The second part of the study consisted of beach-wide longitudinal surveys targeting the upper sand aquifer of the near-shore beach area, which would indicate P concentrations in groundwater just prior to being discharged to the bay and would reveal any broader spatial patterns of groundwater P that might be linked to septic system decommissioning. To start, 10 sites (WA1 to WA10, Fig. 1c; choice of locations were limited to provincial park property) were selected for installation of multi-level wells and for shallow groundwater sampling along the shoreline. Sites spanned from 67th Street to the west (WA1) to 17th Street to the east (WA10). Sewer installation commenced in the early 1980s in the area east of WA7, between the Nottawasaga River and bay, encompassing sites CS-2A, -3A, and -5 and WA7 to WA10. For the areas west of WA7, covering CS-6 and WA1 to WA6, sewers were installed in the block closest to the bay (300–500 m) in 1996–2000 and in the block further inland in 2000–2005. Additional shallow groundwater sampling (but not multi-level well installation) occurred at 2 sites further east of WA10 (WA11, WA12, Fig. 1c), on a narrow strip of beach backed by the Nottawasaga River and that has no existing or decommissioned septic systems. Thus, these last 2 sites may be considered as non-wastewater reference sites (i.e., representing background groundwater conditions).

Methods

Well installation and sampling

A network of multi-level monitoring wells was installed in 2014 at 10 sites (Fig. 1c) along a 6-km section of Wasaga Beach from 67th street in the west (site WA1) to 17th street in the east (site WA10). These were placed a few meters up-gradient of the edge of the permanently raked (vegetation-free) beach zone; their distance from the lake shoreline would typically (i.e., on a calm day) be about 50 m during the summer-autumn periods of this study. Each multi-level well contained 6 to 9 polyethylene tubes (5-mm diameter; 5-cm screen length) bundled around a central PVC pipe (1.2-cm diameter; 10-cm screen length at 10 m) for sampling groundwater at various depths down to 10 m. A direct-push rig was used to install the wells, with cores of the sediment collected for visual inspection during each installation and the surrounding sand allowed to collapse around the well upon drill-rod removal with vibration. Some additional cores were taken at CS-5; these underwent grain size analysis at the Agriculture and Food Laboratory, University of Guelph, with estimates of saturated hydraulic conductivity made using the Hazen method (1911). A small fully-screened 2.5-cm diameter PVC well was installed in front of the septic bed at CS-5, in which the water table was monitored continually from Oct 2015 to Oct 2016 with a Diver pressure transducer (Solinst, Georgetown, ON).

Shoreline profiling

At each of the 10 sites described above plus two additional reference sites (WA11 and WA12; Fig. 1c), a series of shallow groundwater samples were collected just inland from the swash zone (a few meters from the lake), in 2014 and 2016, respectively. Each of the 12 series consisted of eight locations spaced 20 m apart running parallel to the shore, with samples collected from two depths at each location. Samples were obtained with a mini-profiler (Roy and Bickerton 2010), which has internal polyethylene tubing inside a steel rod attached to a stainless steel tip (2.5-cm diameter) with screened sampling ports (5 cm section). It was driven into the sand using a hand-held hammer-drill. Groundwater was first sampled at approx. 50 cm depth; the mini-profiler was then pushed deeper to sample at a depth between 80 and 120 cm generally (deeper preferred but where an adequate flow of groundwater was still available). Lake water was pumped down the mini-profiler tubing with a peristaltic pump during driving to prevent clogging of the ports. Groundwater was then withdrawn using the pump, with sample collection initiated only when field-measured parameters (electrical conductivity (EC), temperature (T), pH, dissolved oxygen (DO)) stabilized. The hydraulic head difference between groundwater at the deeper sampling location and the lake was then determined by attaching the mini-profiler tubing to a potentiomanometer, with an additional connected length of tubing immersed in the lake. These readings took no more than a few min to stabilize. The mini-profiler was removed from the sand for subsequent groundwater collection at the next sampling location. Several grab samples of near-shore lake water were also collected.

Comfort Station septic bed sites

In 2015, shallow groundwater samples were collected down-gradient of the decommissioned septic beds of 4 comfort stations (CS-2A, -3A, -5, -6; Fig. 1c) using a Waterloo profiler system. The sampling procedure was similar to that described above for the mini-profiler, but being larger (4.5-cm diameter), this one required a jack hammer for driving to the desired depth. At each site, sample locations were typically spaced 5–10 m apart in a line parallel to the shoreline, forming a transect of about 60 m length. Groundwater was collected at each sample location from multiple depths, at intervals of 50 cm from 0.3 m to a maximum of 1.8 m below the water table.

Guelph permeameter

Measurements of the in situ field-saturated hydraulic conductivity of the shallow beach sand, i.e., that above the water table, were made in October 2016, using a Guelph permeameter (Reynolds and Elrick 1987; Hoskin Scientific, Burlington, ON). Several measurements were made at each of two beach area sites: at CS-5 (near WA7/8) and at WA11. A previous study (Crowe et al. 2011) made similar measurements at the beach near WA1 and WA2 (Fig. 1c).

Sample handling and chemical analyses

Water samples were filtered (0.45 μm) and preserved as appropriate (see below), and then stored on ice in the field and during transport to either the Canada Centre for Inland Waters (Burlington, ON) or the University of Waterloo (Waterloo, ON). These underwent analyses for major cations (after preservation at pH < 2 with 10% nitric acid), major anions, artificial sweeteners, SRP, and ammonium (after preservation at pH 5–6 with 10% hydrochloric acid), as described previously (Roy and Bickerton 2010; Robertson et al. 2013) and in Electronic Supplementary Material (ESM) Appendix S1. Testing of groundwater SRP concentrations in the field with CHEMets kits (Cole-Parmer Canada) showed good similarity to lab values, suggesting no SRP losses during transport. Select sediment samples from the cores were analyzed for carbon fractions (total, inorganic, organic) at the Agriculture and Food Laboratory, University of Guelph.

Results and discussion

Aquifer conditions and groundwater flow

The surface geology of the Wasaga Beach area (Fig. 1b) consists of lacustrine sand and gravel, which is underlain by a clay aquitard (Johnson, 1999). Cores collected during the installation of the 10 multi-level wells suggest that the near-shore surficial aquifer across the study area is predominantly sand (WA3-WA10), with a thickness increasing westward from <2 (WA3–5) to >20 m (WA10). However, there is also a sand-gravel aquifer with boulders at WA1 and WA2 of >8 m thickness, which is visually apparent along the shore at ground surface and in air photos. Core samples at CS-5 site (near WA7 and WA8) indicated that the beach sand aquifer is comprised predominantly of fine to very fine sand. On average, the matrix is 1.5% gravel, 97% sand, 0.7% silt, and 2.3% clay. Thus, with the shallow sand aquifer apparently quite homogeneous and with no fine sediments overlying it in the near-shore bay, this likely results in relatively diffuse groundwater discharge along the entire beach area close to the shoreline, rather than focused zones and spring-like point discharges as was noted in some areas studied by Kidmose et al. (2013) for a lake with substantial fine sediments over a sand aquifer. The inorganic carbon ($CaCO_3$) content of the sand was typically >1%, with some samples >20%. This fits with the groundwater chemistry being predominantly of calcium bicarbonate type (see below), as was the near-shore lake water (not shown). Relatively high levels of organic carbon (up to 4.0 wt%) were also present in parts of the CS-5 site cores and particularly at a depth of 1–2 m.

Estimates of the saturated hydraulic conductivity (Ks) and field-saturated hydraulic conductivity (Kfs) for the sand aquifer are given in Table 2. The Ks based on aquifer grain size data (CS-5 site core; mean of 6.9 m/d) were similar to Guelph permeameter measurements made 30–50 cm below the sand surface (i.e., above the water table; CS-5 site and beach area near WA11; mean of 3.1 m/d), considering that Kfs is typically two or more times lower than the actual Ks (Marinas et al. 2012 and references therein). The full range of Ks data ranges 3 times about the mean. Crowe et al. (2011) used this same Guelph permeameter at their Wasaga Beach study site, which was in the sand-gravel aquifer near WA1 and WA2, and reported Kfs ranged from 15.5–36.7 m/d, with a geometric mean of 21.9 m/d.

The elevation of the water table determined periodically at the 10 multi-level well sites during the summer and autumn periods was always higher than that of the lake (lake level not measured precisely), indicating that groundwater was flowing toward the lake at these times. A similar result was found by Crowe et al. (2011) for areas around WA1 and WA2 in autumn. Furthermore, potentiomanometer measurements at the shoreline (just above the swash zone) indicated that groundwater was discharging to the lake, based on a positive head difference ($dH =$ groundwater head - lake head) for WA11 ($dH = 3.0$ cm; Oct 2016) and WA1–10 ($dH = 3$–38 cm, generally lower for eastern sites; Oct 2013). Also, during wet periods there were visible signs along the entire beach of discharging groundwater, including iron staining at the sand surface. The water table well at CS-5, monitored from Oct 2015 to Oct 2016, showed a high water table in winter through early spring, with a general decline to its low point at the end of summer into early autumn, with a total variation of ~0.5 m. In comparison, the lake level from late May to early Oct, as reported from an ECCC buoy stationed nearby in Nottawasaga Bay, had a stable long-term average (with occasional short-term storm-related variations of <0.5 m occurring over several days). This suggests the hydraulic gradient of the beach groundwater mimicked the seasonal pattern of the water table (during the non-frozen period), with higher flows in the spring compared to the late summer – early autumn.

The CS-5 well and lake level data allow for a calculation of the hydraulic gradient (dH/dL) at that site. With the well typically 41 m from the lakeshore and the head difference varying throughout the monitoring period from 0.3–0.8 m, dH/dL varied from -0.0073 to -0.020 (note negative value, because of assuming distance, L, increases from inland toward the lakeshore). Crowe et al. (2011) determined the dH/dL for their study area, by WA1 and WA2, in the autumn of 2009. Using measurements of the water table at distances of 3 and 45 m

Table 2
Estimates of saturated (Ks) and field-saturated (Kfs) hydraulic conductivity for aquifer materials at Wasaga Beach made using the Hazen (1911) method (HM) using grain size data or using a Guelph permeameter (GP; Reynolds and Elrick 1987). All estimates are from this study except GP near WA1 & WA2, which were reported by Crowe et al., (2011).

Method and location	No. samples	Range (m/d)	Mean (m/d)
HM on sand cores fromCS-5	33	Ks, 6.0–12.1 (59.5 outlier)	Ks, 6.9
GP at CS-5	5	Kfs, 1.0–2.8	Kfs, 3.1
GP near WA11	3	Kfs, 2.7–8.3	
GP near WA1 & WA2	3	Kfs, 15.5–36.7	Kfs, 21.9

from shore along a line perpendicular to the shoreline, they measured a head difference of 0.7–0.8 m, which gave dH/dL of −0.017 to −0.019.

Groundwater velocity (v) can be calculated for the CS-5 site using Darcy's law:

$$v = -\frac{Ks}{n} dH/dL \tag{1}$$

where n is porosity. Crowe et al. (2011) measured porosity of the sand as 0.40 at a beach north of their field site. However, the beach sand near the shore at CS-5 is quite compact, perhaps in part due to many decades of vehicle traffic, so we assumed a value of 0.35. Using a range of Ks of 2 to 17 m/d (Table 2) and a range of dH/dL of −0.0073 to −0.020, results in a groundwater velocity of 0.04 to 1.0 m/d towards the lake. Our best guess for the most representative conditions would have Ks of 7 m/d and dH/dL of −0.015, giving a velocity of 0.3 m/d towards the lake. In comparison, Crowe et al. (2011) calculated a groundwater velocity of 0.85–1.0 m/d toward the lake for the sand-gravel beach area near WA1 and WA2. This higher velocity likely reflects a greater Ks for the sand-gravel aquifer. On a final note, the shoreline head difference measured by potentiomanometer was typically less (3–4 times) for sites further east (i.e., WA9–WA11; WA12 not done; Fig. 1c). Thus, groundwater velocity may be less in these areas, perhaps due to the greater depth of the aquifer to the east.

Site-scale study component

P plume occurrence

Groundwater sampling at 4 provincial park comfort stations (Fig. 1c; Table 1) revealed distinct groundwater P plumes, indicated by a localization of elevated concentrations of SRP, at sites CS-5 and CS-6, but not at CS-2 and CS-3 (Fig. 2). Each of the two P plumes detected was directly down-gradient of its tile bed area. These P plumes were not accompanied by similar plumes of elevated concentrations of common wastewater tracers (ESM Fig. S1–4), including nitrate, ammonium (not shown; <0.8 mg/L), chloride, and artificial sweeteners, or any notable difference in major ion water chemistry. Thus, these are clearly legacy P plumes associated with the decommissioned septic systems of CS-5 and CS-6 rather than plumes resulting from a current or recent source of wastewater contamination (e.g., leaky sewer).

The presence of the P plumes at CS-5 and CS-6, where the septic systems were decommissioned 34 and 14–18 years previously, respectively, illustrates their potentially long persistence (decades). However, it is not obvious whether the lingering P plumes have been sourced from existing P retained on the aquifer sediment and/or from leaching of P formerly held in the vadose zone. For the latter situation, reduced water infiltration following the loss of liquid inputs to the septic system would likely result in a vertically-thin plume at the top of the capillary fringe, though dispersion due to water table fluctuations could act to

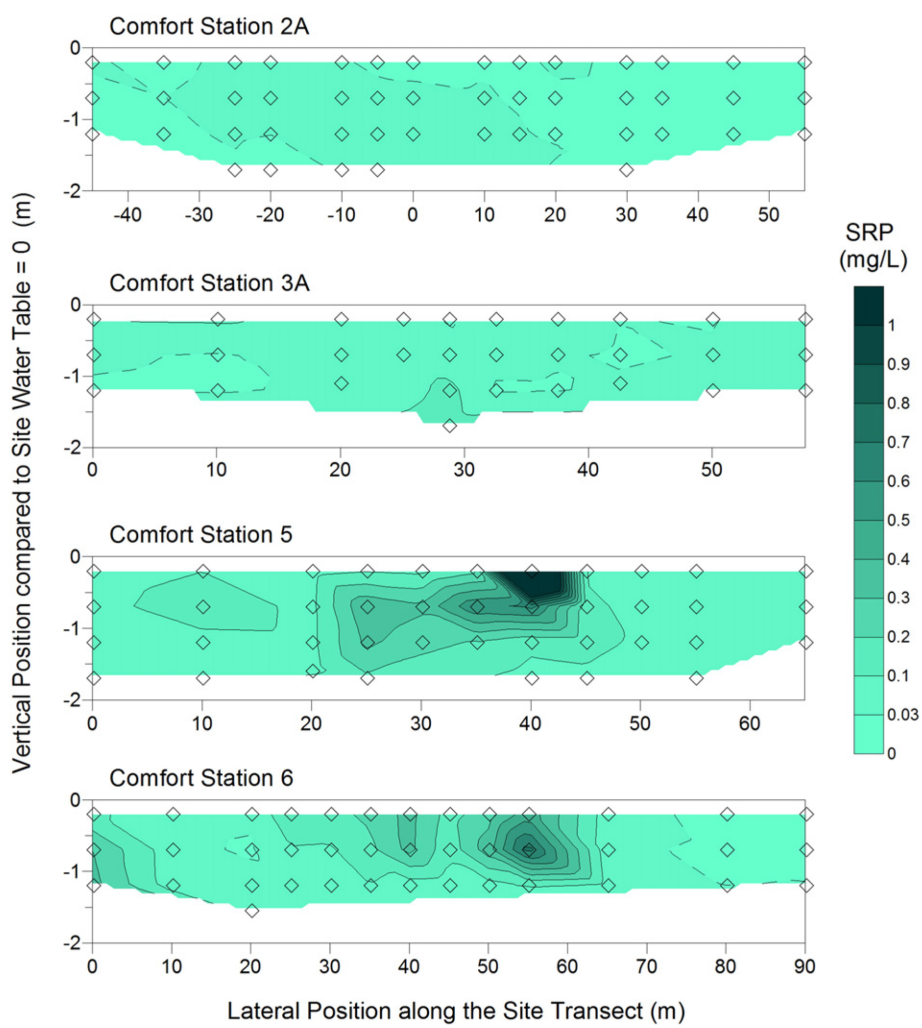

Fig. 2. Interpolated plots of soluble reactive phosphorus (SRP) concentration from groundwater samples collected directly down-gradient of the four decommissioned comfort station septic beds. Plots are cross-sections (parallel to the shoreline): depth (y) below the water table vs. distance along a transect perpendicular to groundwater flow (x). Symbols indicate sample position; interpolation by linear triangulation using Surfer (Golden Software, Inc., Colorado). Maximum concentration was 2100 μg P/L for Comfort Station 5.

create a thicker plume and this cannot be ruled out here. Meanwhile, plumes emanating from active septic systems (i.e., greater recharge) tend to be vertically-thicker, like these 2 plumes (Fig. 2), which suggests that these are sourced primarily from the remnants of the old septic groundwater plumes, and remain due to high sorption capacity of the aquifer material for dissolved P (Robertson and Harman 1999; McCobb et al. 2003; Parkhurst et al. 2003). A similar argument was made by Robertson and Harman (1999) for a P plume monitored up to 4 years post-decommissioning.

These findings show that dissolved P from septic systems can reach the water table and form plumes of P in this sand aquifer. This fits with observations of septic-sourced P plumes being present in groundwater with calcareous sediments despite their absence in non-calcareous sediments (Robertson et al. 1998; Robertson 2003). Of note though, these previously-reported P plumes occurred under oxidizing conditions, whereas at Wasaga Beach the groundwater was strongly reducing (low DO, low nitrate, elevated iron; Fe-reduction redox zone).

There are several possible explanations for why P plumes were not detected at CS-2A and -3A, even though their septic systems were decommissioned around the same time as that of CS-5 (i.e., 30–35 years previously), where a remnant P plume was found. It's possible that P plumes never formed at these comfort stations. However, given the similar sediment (beach sand) and groundwater chemistry (predominantly calcium bicarbonate water type and pH = 6.5–8.0; ESM Figs. S5–8, with generally low DO (<2 mg/L)), and with similar depths to the water table (CS-2A – 70–90 cm; CS-3A – 80–90 cm; CS-5–50 cm; CS-6–120 cm) between the 4 comfort station sites, there is no strong reason to expect a stark difference in P transport to groundwater (i.e., via enhanced P mineral precipitation in the vadose zone). Alternately, these two sites may have had P plumes that have been flushed past the sampling transect, especially if it were closer to their respective plume sources than for CS-5 (as for CS-2A, Table 1). Other possible reasons include having a higher groundwater velocity (not measured), lower initial P mass in the aquifer (no information on P loadings available), or lower sorption capacity of the sand (not measured) compared to the CS-5 site. Finally, it may be that the plumes have travelled deeper than our maximum depth of sampling (i.e., >2 m below the water table), as was the case for Rea and Upchurch (1980), and are still present at the distance of the sampling transect. Some smearing of the plumes to shallow depths would likely have occurred, though, especially post-decommissioning, which might have been apparent in these P cross-sections (Fig. 2a,b). Considering the above arguments, we suspect that the plumes have been flushed out beyond the profiler transect, but the uncertainty around these missing plumes will not overly affect the calculations of the following sections, being based largely on the CS-5 and CS-6 plumes.

P plume travel model

Although temporal data for these 4 comfort station sites are lacking, some estimates of limits on the P plume velocity can be made. Here we consider P retardation via sorption only; long-term entrapment with the precipitation of P-bearing minerals (e.g., vivianite) may be occurring, and would lead to lower apparent velocities. Assuming the P source to the plume was stopped upon decommissioning, the presence or absence of a plume at the profiling transects can provide some bounds on how quickly the trailing edge of the plume is being flushed from the aquifer. Here we assume 1-D plug flow (horizontal flow, advection only).

The up-gradient extents of the P plumes (i.e., starting position) upon decommissioning are not known for certain. However, estimated minimums or ranges of this were made based on visual surface assessments of the septic bed locations, with the assumption that the groundwater plume formed somewhere within the septic bed footprint, knowing that such plumes may not coincide with the full dimensions of their septic bed (Robertson, 2012). From this, the distances between the initial P plumes and the profile transects have been estimated (Table 1). These are more constrained for CS-5 and CS-6 because the location of the septic bed was more evident. Considering this distance and the estimated years passed since decommissioning, a maximum velocity limit was determined for sites where P plumes still existed, i.e., for CS-5 and CS-6, while a minimum velocity limit was calculated for CS-2A and CS-3A (Table 1), assuming shallow P plumes existed but have been flushed out.

Plume velocity depends on the groundwater velocity, v, and the sorption properties of the aquifer sand at each site. However, it is interesting to note that the general limits for the 4 sites (Table 1), if we assume similar conditions at each site, suggests a P plume velocity ~1 m/a, which is similar to values reported for other septic plume sites in shallow calcareous sand aquifers: Cambridge, ~1 m/a (Robertson 2003; v = 24 m/a), Camp Henry, 1 m/a (Ptacek 1998; v = 10–60 m/a), Long Point, 0.8 m/a, (Robertson 2008; v = 28 m/a), Langton, 1.7 m/a (Harman et al. 1996; v = 100 m/a). Now if linear (fast and reversible) sorption isotherm is assumed, then we can determine the retardation factor (R = average groundwater velocity/P plume velocity) for P of these septic plumes. The average v was estimated (above) as 110 m/a (0.3 m/d) for site CS-5; considering its P plume traveling at most 1.3 m/a, then $R \geq 85$. If we assume the average v has been similar across the entire Wasaga Beach, then for CS-6, with a P plume traveling at most 1.8 m/a, then $R \geq 60$. For CS-2A and CS-3A, if plumes existed but were flushed out, the plumes traveling >0.6 and >1.1 m/a, respectively, results in $R \leq 180$ and ≤ 100, respectively. In a review of P transport associated with 10 septic plumes in unconsolidated sediments, Robertson et al. (1998) reported R generally ranged from 20 to 100, with 1 site not determined. The majority of those sites were under oxidizing groundwater conditions, unlike the reducing conditions predominating for all 4 Wasaga Beach CS sites. Still, this reported range is in general agreement with these Wasaga Beach estimated limits.

We now continue with a rough extrapolation (outlined in Table 3) of the comfort station site results for septic P plume transport to the whole of the decommissioned septic systems of the town of Wasaga Beach (Fig. 1c). The closest block of cottages (east and west ends) typically starts at 50–75 m from shore (though in some areas these closest cottages have been replaced by park land and roads) and ends 250–500 m from shore. Meanwhile, the further block of cottages extends several km from the shore, though on the east end, the Nottawasaga River likely intercepts P plumes sourced from this further block. Historical records indicate that some hotels and cottages were present at Wasaga Beach, mostly along the beach front itself, by the turn of the 20th century. Streets and cottages of the 'beach strip' (we suspect this means the closest block of cottages) were well-established along the length of Wasaga Beach by the 1950s. Census data from Statistics Canada (1981) state that there were 220 private dwellings constructed

Table 3
Summary of P plume transport calculations applied to the decommissioned (retired) septic systems of the town of Wasaga Beach. Travel years = current year – year septic system built; flushed years = current year - year septic system retired; current year = 2015.

	Year system built or retired	P sorption (this study)	P sorption (alternate)
Groundwater velocity, v (m/a) (eq. 1)		110	
Sorption retardation coefficient[a], R		85	20
P plume velocity, v_p (m/a) = v ÷ R		1.3	5.5
P plume travel extent (m)	1900	150	630
$=v_p \times$ travel years	1950	85	360
	1980	46	190
P plume flushed distance (m)	1980 (east end)	46	190
$=v_p \times$ flushed years	2000 (west end)	20	83

[a] Study retardation coefficient was derived from apparent P plume velocity limits from the comfort station data and the determined groundwater velocity, 110 m/a); alternate (lower) retardation coefficient from literature reports.

prior to 1946, with >1000 added by 1970, with a total of 1930 in 1981. We assume the timing of septic systems installations coincided with that of cottage development.

Information on v and P sorption properties for Wasaga Beach is quite limited, so we will assume that the P plume velocity of 1.3 m/a determined from CS-5 applies throughout the cottage and beach areas along the lateral extent of Wasaga Beach, and hasn't changed substantially over the past century. This gives a maximum P plume travel extent (Table 3) of 150 m for systems built in 1900 (115 years of travel, to 2015), and a range of 46–85 m for those built between 1950 and 1980, which is the majority of the dwellings. This applies to plumes not intercepted by discharge to a river, ditch, or the lake; taken up by vegetation; or affected by P mineral precipitation (note that town supply wells are not in these cottage areas). These short distances highlight how slow the P plumes may be travelling to the lake. Even if we consider $R = 20$, a lower value common in the literature for calcareous sand, which would result in P plumes moving at 5.5 m/a, the majority of plumes would still only travel a few hundred m (Table 3). Given these crude calculations, it seems likely that septic P plumes from the closest cottages (50–75 m from shore) would have reached the shore (unless R is much >85). Plumes from the furthest set of cottages in this first block (i.e., >360 m from shore; and likely built after 1950) are less likely to have reached the shore. Meanwhile, P plumes from cottages that are still further away are probably not close to shore yet.

The plume flushed distance, that travelled by the back end of the plume since decommissioning, can be calculated similarly (Table 3). For the east end (decommissioned 30–35 years), this distance is at most 46 m ($R = 85$; or 190 m for $R = 20$); for the west end

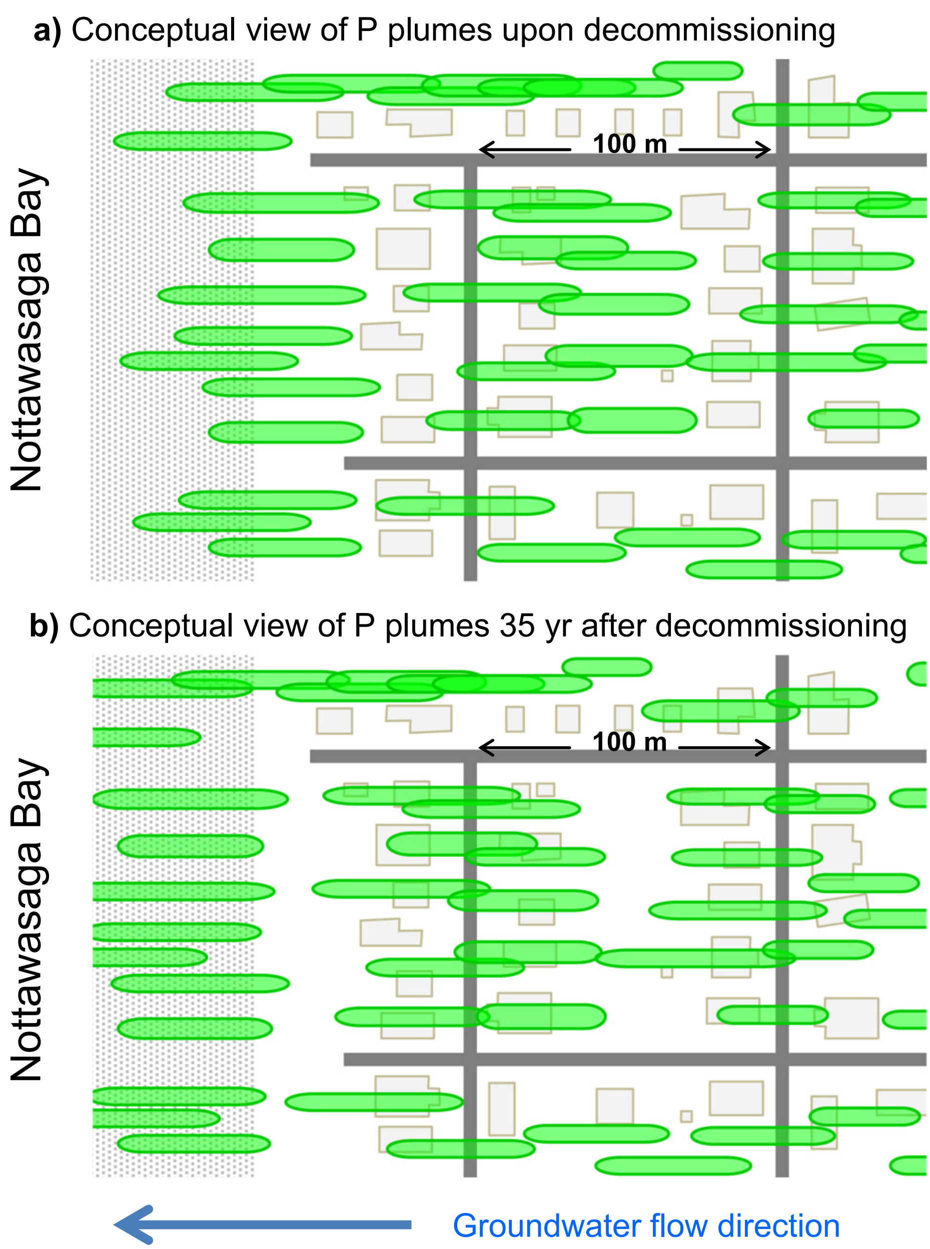

Fig. 3. Plan-view diagrams illustrating the conceptual model for septic P plumes (elongated green tubes; width dependent on septic bed size and functionality; ignores dispersion) as 'slugs' migrating slowly in the groundwater from the residential area (black lines are roads, buildings as rectangular grey shapes) toward the beach (stippled area) and Nottawasaga Bay. The situation depicted in diagram a) represents the time of decommissioning (here 1981), with the in-shore end of each plume originating at the septic bed (assumed placement in backyard of each property, including from demolished dwellings) and the plume length indicating the time passed since septic system installation. Diagram b) represents the current situation, ~35 years later, with P plume slugs having travelled ~50 m towards the shore.

(decommissioned 15–20 years), it may only be 20 m (R = 85; or 83 m for R = 20). The closest block of cottages is unlikely to have been flushed out for either section; indeed it may be that none or only very few of the very closest plumes (cottages 50–75 m from shore) have been flushed out (if $R < 85$ in those areas or v is >110 m/a). Certainly the plumes for CS-5 and CS-6 comfort stations are not even close to being flushed out, and their septic systems are about as close to the shore (~50 m) as any other septic system could be. These calculations suggest it may take many more decades, or up to a couple of centuries (R = 85), to flush from the aquifer any P plumes produced from septic systems of the closest block of cottages, and longer still for the further block of cottages.

These P plumes may be visualized as a throng of slow moving 'slugs' migrating from their source septic systems toward the bay (Fig. 3), as most are unlikely to have yet reached the bay. Assuming homogeneous P retardation in the aquifer and similar groundwater flow conditions, the length of each plume (slug) will depend solely on how long the septic system was functioning (i.e., construction date until decommissioning date). An average septic system life of 50 years would give a plume length of 65 m (R = 85), with the majority of any longer plumes likely originating from cottages closest to the beach, having been constructed earlier. Those plumes closest to the bay will reach it and be flushed from the aquifer sooner, while those from further blocks of cottages will arrive and be flushed out later, perhaps by centuries. Indeed, the full complement of P plumes will not be at the shoreline together at any one time (Fig. 3), meaning P loading from the town's septic systems will be spread out (dispersed) over a long period of time. Right now, we suspect that the P plume loading is still on an increasing trajectory, with new plumes arriving, but few plumes being flushed out.

Beach-scale study component

Multi-level monitoring wells

The first beach-wide survey of groundwater consisted of 10 multi-level wells (Fig. 1c), which covered ~6 km of beach. The limited spatial coverage provided by the single multi-level well per site means they may not distinguish individual P plumes, should they exist. However, the potential mixing of multiple plumes from the many hundreds of septic systems, along with effects of dispersion, may allow these wells to capture some general patterns. The depths of sampled groundwater covered by each well (Fig. 4) represent the thickness of the surficial sand aquifer (for those <8 m) because few samples were obtained from the underlying clay aquitard.

The groundwater dissolved oxygen was consistently low, indicating that anoxic conditions prevailed throughout the sampled profiles at all sites. Ammonium and nitrate concentrations (not shown) were <1.5 and <1.7 mg/L, respectively, with no spatial patterns evident. The SRP concentrations varied between 2 and 240 μg/L and generally decreased with depth (Fig. 4). They are much lower than those found in the main body of the P plumes detected at CS-5 and CS-6 (typically >300 μg/L, up to 2100 μg/L; Fig. 2), which suggests a general absence of P plumes

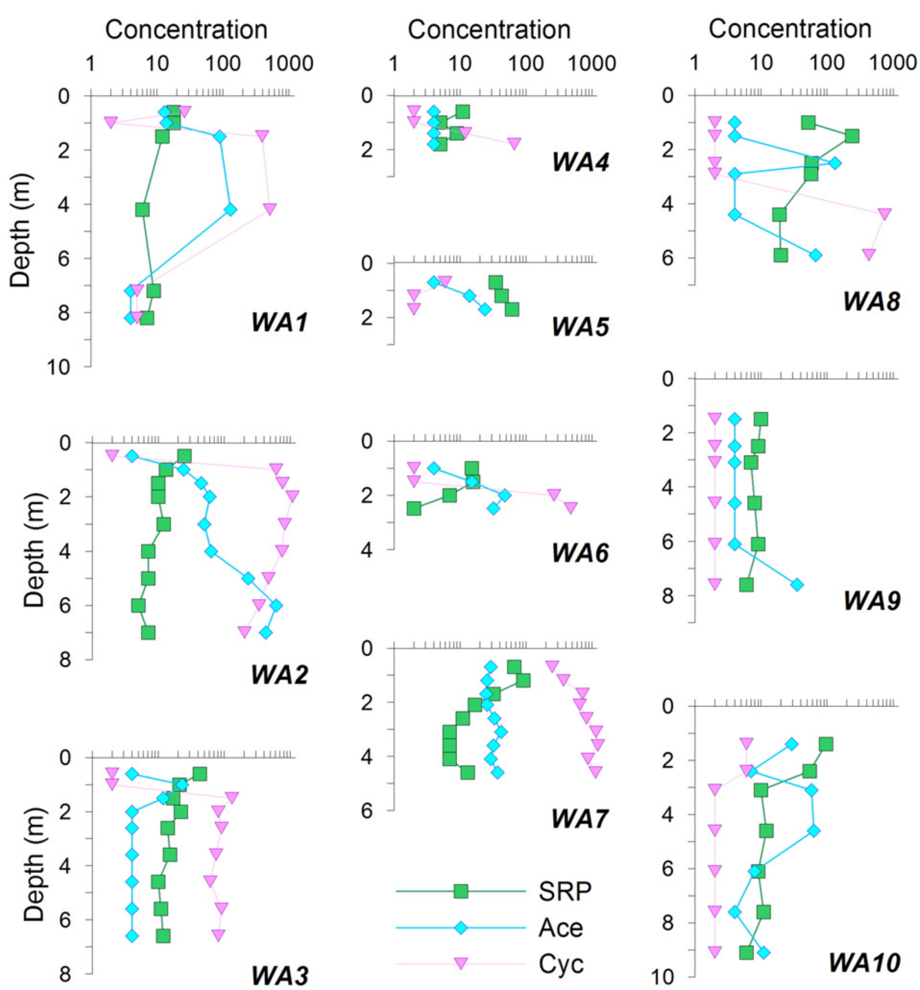

Fig. 4. Depth profiles of groundwater concentrations (symbols) for soluble reactive phosphorus (SRP; μg/L) and the artificial sweeteners (ng/L) acesulfame (Ace) and cyclamate (Cyc), for the 10 multilevel wells at sites WA1 (west) to WA10 (east), as shown in Fig. 1c. Non-detections for Ace and Cyc are plotted at the minimum detection limit concentration (4 and 2 ng/L, respectively). Lines connecting symbols are just a visual aid.

across all of the sites. This may reflect very slow P plume velocities, as was found for plumes at CS-5 and CS-6, meaning few plumes have reached the beach yet. Alternately, there could be a lack of P plumes forming in the cottage zones off the beach, if conditions differ from those of CS-5 and CS-6, though we have no good reason to suspect that is the case. Finally, it may simply reflect poor capture by the multi-level wells of septic-sourced P plumes. The cause of the higher SRP at shallow depths (Fig. 4) may be other sources of P occurring at the land surface (e.g., fertilizers, bird droppings) or within the soil and shallow aquifer itself (e.g., sediment minerals or decaying organic matter; noting the higher organic matter content at 1–2 m depth in cores from CS-5).

In comparing the SRP concentrations from the two decommissioning time zones (i.e., WA1–6 vs. WA7–10; Fig. 1c), the east zone (earlier decommissioning) has a mean value twice that of the west zone (34 vs. 15 μg/L), though the median values are the same (12 μg/L). The lack of any trend with decommissioning fits with the slow P plume transport scenario derived in the section above, though it also might reflect poor capture of P plumes by the multilevel wells.

The artificial sweeteners measured here, being anionic, tend to act as conservative (non-sorbing) solutes, with Ace and Suc being more recalcitrant than Sac and Cyc in many conditions (Lange et al. 2012). All ten wells had detectable concentrations of three of the artificial sweeteners (Ace, Sac, Cyc), with some samples having elevated concentrations (e.g., >100 ng/L) of Ace and Cyc that are strongly suggestive of an influence of wastewater, either from decommissioned septic systems or leaky sewers. There were no trends for the sweeteners with decommissioning time zones, with much variability from site to site. However, in contrast to the depth-trend for SRP, the sweetener concentrations were typically

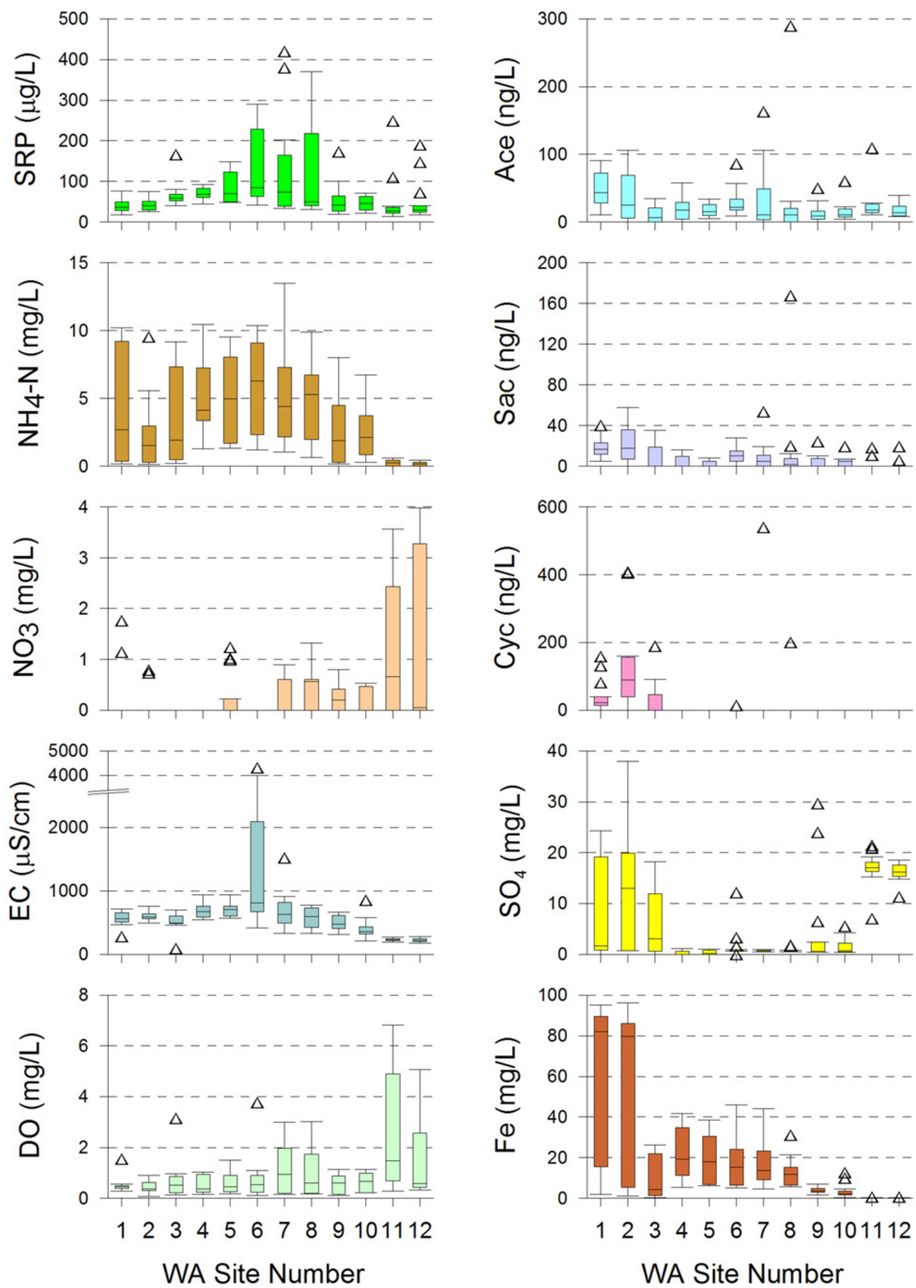

Fig. 5. Box-whisker plot of shallow groundwater chemistry data (SRP – soluble reactive phosphorus, NH_4-N – ammonium as N, NO_3 – nitrate, EC – electrical conductivity, DO – dissolved oxygen, Ace – acesulfame, Sac – saccharin, Cyc – cyclamate, SO_4 – sulphate, Fe – iron) from shoreline profiling in 2013 (WA1–10) and 2016 (WA11–12). Two sampling depths (~50 cm and 80–120 cm) combined. Box indicates 25th–75th percentile, with median line internal; whiskers show data limits excluding outlier data (triangles, with outlier factor 1.5 of interquartile range).

low at the shallowest sample point(s), with higher concentrations (Ace, Cyc only) at some greater depths (Fig. 4). Indeed, there was poor linear correlation ($R^2 < 0.13$) between SRP and any of these sweeteners. This SRP-sweetener concentration depth pattern (Fig. 4) may reflect the interacting factors of distance of the septic system from shore and differing transport velocities of the SRP and sweeteners. Groundwater affected by septic systems nearer the shore will likely be found at shallow depths while the influence of septic systems further from the shore is likely to be detected at greater depths, due to the addition of recharge water from surface all along the groundwater flow path towards the shore. Sweeteners from decommissioned septic systems near to shore have likely been flushed past the monitoring well (and out to the bay), given a probable high v ~100 m/a, while the corresponding P, likely sorbing to the aquifer sediments (as noted above and in the literature, e.g., Parkhurst et al. 2003), may still be slowly transported in the shallow part of the aquifer (potentially ~1 m/a, noted above). For septic systems further from shore, their sweeteners may still be present in the deeper portion of the aquifer near the shore while the corresponding P has not yet reached the beach area. The general lack of elevated concentrations of acesulfame, which has been in use in Canada for only about two decades (Gougeon et al. 2004), in comparison to saccharin and cyclamate, which were introduced over a half-century ago, is suggestive of groundwater affected predominantly by 'old' wastewater (as fits with old septic plumes). There was only 1 well with notably elevated acesulfame at greater depth than where the peak in cyclamate occurred (WA2: >0.6 μg/L; Fig. 4). This well was in the potentially more permeable sand-gravel aquifer, which may have allowed faster groundwater transport of sweeteners from the cottage block most recently decommissioned (2000–2006; Fig. 1c) further from the shoreline.

Shoreline profiling

The next beach-wide investigation involved sampling shallow groundwater along the lakeshore (i.e., close to its discharge point) using a profiling series (8 locations, 2 depths: ~50 cm and 80–120 cm) at each of 12 sites (WA1–10 adjacent to the developed portion of town; WA11–12 as 'reference' sites; Fig. 1c). Comparison of the SRP and other groundwater chemistry results (Fig. 5) between these different groups of sites may provide information on the sources of P to this groundwater, which may include the town's decommissioned septic systems, and any influence of the time passed since decommissioning on the spatial pattern of groundwater P. However, these shallow groundwater samples, though from above the swash zone as it existed that day, might be influenced by lake water infiltration from wave run-up during short periods of higher lake levels (i.e., storm events). This would tend to dilute the concentrations of SRP and other chemical parameters. A comparison of water chemistry indicates that these samples (Fig. 5) fall within the broad ranges found for groundwater of the multilevel wells and comfort station profiling (Table 4), and that the majority tend to differ markedly for several parameters from samples of near-shore lake water (especially for EC, DO, Fe, SO_4; though many of these may change with time spent in the sand). The reference

Table 4
Ranges of water chemistry values for near-shore lake water (3 samples) and inland beach aquifer from sampling multi-level wells and profile sampling the 4 comfort stations.

Parameter	Lake	Wells	Profiling
EC (μS/cm)	150–250	340–2800	125–890
DO (mg/L)	>10	<3	<6.7
Chloride (mg/L)	8–14	2–1000	1–57
Nitrate (mg/L)	1–1.5	0–1.7	0–7.3
Ammonium (mg/L)	0.02–0.23	0–1.5	0–0.77
Sulphate (mg/L)	12–15	0.1–49	0.6–79
Iron (mg/L)	0.002–0.2	0.6–12	0–17
SRP (μg/L)	6–28	2–240	8–2000
Acesulfame (ng/L)	7–12	7–610	6–880

samples seem most similar to the lake water, and may be diluted in SRP somewhat. Still, lake water infiltration seems likely only minor for the majority of the shallow groundwater samples.

Although the concentrations of artificial sweeteners were generally low (Fig. 5) compared to levels expected in wastewater plumes (1000s of ng/L), there was a pattern of elevated average concentrations of all 3 artificial sweeteners at the 3 sites at the west end of Wasaga Beach, where septic systems were most recently decommissioned (2000–2005, in a block 1–2.5 km from the beach; Fig. 1c). This would suggest an average groundwater velocity of <250 m/a, assuming the artificial sweeteners travel conservatively in the aquifer (as expected; Lange et al. 2012). This velocity limit fits with the velocity estimates derived above. Concentrations of Fe and SO_4 were also elevated at these sites, while those of NO_3 were below detection (Fig. 5); this may signify nitrate reduction with pyrite, as has been reported for septic system nitrate previously (Aravena and Robertson 1998. This spatial pattern indicates that groundwater affected by distant septic systems can be captured with this shallow profiling method, at least in some areas. It also suggests that the conservative solutes of the decommissioned septic plumes from the other areas (i.e., WA4–10) have been flushed out of the aquifer, as expected based on the v calculations.

The individual median SRP concentrations for the 12 sites ranged from 27 to 84 μg/L (Fig. 5). The reference sites (WA11–12) had the lowest median SRP concentrations, but were also notable for having more oxygenated groundwater conditions (i.e., higher DO, NO_3, SO_4; with lower NH_4-N and Fe; Fig. 5), whereas the vast majority of samples from the other sites were under Fe-reducing conditions (low DO, NO_3; high Mn, Fe concentrations; Fig. 5). All five reference site samples with SRP > 50 μg/L were under Fe-reducing conditions (not shown), which suggests the oxidative redox state limited the SRP concentrations for the reference sites. This is not surprising since iron oxides are known to bind P and therefore reduce its dissolved concentration and mobility in the subsurface. These oxic conditions might be linked to lake water infiltration, as discussed above. Despite this effect, there was no significant difference in SRP concentrations between sites WA1, WA2, WA3, WA9, WA10, WA11 and WA12, as determined by a Tukey test with one-way ANOVA – (log-transformed; residuals were approximately normally-distributed). This finding suggests that these 5 non-reference sites, at least, have not necessarily experienced elevated SRP concentrations due to the influence of septic P plumes.

There is no clear difference in SRP concentrations (Fig. 5) between the east (WA1–6) and west (WA7–10) decommissioning zones (Fig. 1c), as fits with the plume model results suggesting it will take many decades still to flush P plumes from the beach aquifer, regardless of the decommissioning date. The majority of shallow groundwater samples at all WA1–10 sites had SRP < 100 μg/L, though several sites (e.g., WA5, WA6, WA7 and WA8) from each zone had a greater number of samples with SRP concentrations >100 μg/L. Maximum concentrations for septic plumes typically reach 1000–5000 μg/L (e.g., Ptacek 1998; Robertson 2003; McCobb et al. 2003), though the spatially sparse samples along the Wasaga Beach shoreline may not capture maximum concentrations at the heart of a P plume. So perhaps this higher proportion of elevated SRP concentrations may indicate an influence of septic-sourced P plumes at these sites. However, each reference site had a few samples with SRP concentrations >100 μg/L as well, indicating the existence of some non-wastewater source(s) of elevated SRP in the shallow beach groundwater. Maximum concentrations of SRP in groundwater sourced from geologic materials have reached 1300 μg/L (Schilling and Jacobson 2008), 3100 μg/L (Banaszuk et al. 2005), 950 μg/L (Carlyle and Hill 2001), 2900 μg/L (Kelly et al. 1999), 1400 μg/L (Spruill et al. 1998) and 1300 μg/L (Shaw et al. 1990). Thus, the high SRP concentrations at the non-reference sites might also be from a natural source.

Evidence of the interception of a septic P plume in the profiling samples comes from WA7, which crosses directly in front of the CS-5 site. The P plume captured in the CS-5 transect (Fig. 2) aligns with profile locations 6 and 7 (of 8), which are 100–120 m from the start (0 distance,

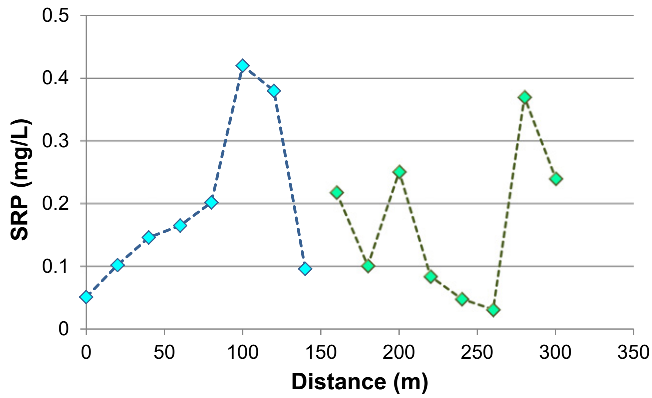

Fig. 6. Concentrations of SRP in shallow groundwater (~80–120 cm depth) collected from the shoreline transects (Fig. 1c) WA7 (distance 0–140 m) and WA8 (distance 160–320 m). Each transect includes 8 locations, spaced 20 m apart. The plume from CS-5 aligns with distance 100–120 m (part of WA7 series).

west end) of the WA7 series as plotted in Fig. 6. This peak in the SRP concentrations (380 and 420 μg/L) indicates that the CS-5 septic P plume has reached the shoreline, a distance of ~50 m from the septic bed. Subsequent sampling (not reported here) has confirmed that the CS-5 P plume has reached the shoreline at this location. The installation date for the CS-5 septic system is not known, but historical records indicate that the province took over control of the beach in the late 1950s, and added change huts and washrooms along the beachfront by 1959. This suggests a maximum time for P plume travel of ~55 years, which results in a minimum P plume velocity of 0.9 m/a. This fits well with the P plume velocity estimate (<1.3 m/a) derived above (Table 1), and supports its supposition that P plumes from the closest set of cottages are likely to have reached the shoreline.

The SRP concentrations of shallow groundwater collected at the adjacent profiling series WA8 are also shown in Fig. 6. Another similarly-high peak in SRP (370 μg/L) occurs at distance 280 m from the west end of WA7. This peak is not associated with the CS-5 comfort station, but may signify a different septic P plume or some other P source. It is unlikely to be from a recent and nearby sewer leak, though, given the very low artificial sweetener concentrations at that and adjacent locations of WA8 (not shown).

The SRP concentrations of the shallow shoreline groundwater were poorly correlated with artificial sweeteners and many other wastewater tracers (e.g., NH_4-N, NO_3, EC), with all $R^2 < 0.06$ (linear regression). Higher concentration outlier samples of these tracers (Fig. 5) rarely coincided with high SRP concentrations, or even with each other. Thus, it is unlikely that nearby leaky sewers (if present) are affecting the beach groundwater or contributing to the elevated SRP concentrations there. Rather, other non-wastewater sources must have contributed to these tracer concentrations. For example, the high EC at WA6 was associated with a narrow dissolved salt (NaCl) plume with low artificial sweeteners, so it was likely sourced from salt deicer use on roads or parking areas. A possible alternate source of high concentrations of individual artificial sweeteners would be food and beverage spills or direct urination on the beach; fittingly, outlier concentrations tended to be at the shallower depth (not shown).

Considering the above analysis, we conclude that the spatial pattern of SRP from the shoreline sampling (Fig. 5) is suggestive of background groundwater concentrations around 30–50 μg/L, with some higher values associated with interception of a septic plume or some other point source of P. This raises two questions: why is there not a greater impact from septic plumes at the shoreline and what is the source of the background SRP? The shallow groundwater sampled at the shoreline is more likely to have been affected by those decommissioned septic beds closest to the shore. Septic plumes sourced from further distances may be too deep to be captured with this method, though the artificial sweetener pattern discussed above suggests otherwise (in some locations at least). Based on the P plume transport model results, such 'near-sourced' plumes are more likely to have travelled far enough to reach the shore, though they may be few and well-spaced. The lack of apparent P plume impacts at the shoreline does fit with the model results that P plumes from septic systems built around the 1950s may have travelled <100 m (Table 3), and thus few of them from the cottage areas would have reached the shore already. It's also possible that P plumes are not forming in cottage areas further from the beach or in other parts of town away from CS-5 and CS-6, or that the P plumes are being captured along the way, by trees or ditches, or attenuated by P mineral precipitation (e.g., forming vivianite with iron). Future groundwater exploration within the town would be needed to fully assess this.

As for the source(s) of P making up the background concentrations in this shallow groundwater, there are several possibilities. These include degrading organic matter in soils, the original aquifer materials, or the reworked beach sands, which are known to entrain organic detritus. Inputs from bird droppings, from the many gulls, geese and ducks that frequent the beach area, may also come from surface, especially at the beach itself. Both are also a supply of ammonium, and indeed the shallow groundwater had high ammonium concentrations (i.e., >1 mg/L) at all sites except for the reference sites (Fig. 5), which had high concentrations of nitrate (likely derived from the oxidation of ammonium, considering the oxic groundwater conditions there). Minerals of the soil or aquifer sediments might also contribute to the groundwater P.

Groundwater P concentrations comparison

The SRP concentrations in the shallow groundwater at the edge of the bay ranged from 13 to 420 μg/L (Fig. 5), with overall median and mean concentrations of 51 and 71 μg/L, respectively. Only 17% (32/189) of the samples had SRP >100 μg/L. These SRP concentrations are generally similar to those in samples from the shallow depths of the multi-level wells (Fig. 4).

These SRP concentrations are only slightly higher than the background groundwater concentration of 30 μg/L reported in the U.S. Geological Survey's National Water-Quality Assessment (NAWQA) Program (Dubrovsky et al., 2010). Their background concentration was derived from 166 wells from areas with minimal human development, but average concentrations were generally similar across the complete nutrient data set of 5101 monitoring, domestic, and public-supply wells, for 3 land-use categories: urban, agricultural, and major aquifers (much deeper wells). Few (12%) of those well samples had dissolved P concentrations >100 μg/L.

These beachside concentrations are only slightly less than found in a pair of studies of urban groundwater discharging to surface waters in Canada. Roy and Bickerton (2014) sampled riparian groundwater along 6 different urban streams; 27% of the 665 total samples had SRP concentrations >100 μg/L. Roy and Malenica (2013) collected 172 samples of shallow groundwater from the shoreline of Lake Simcoe in the city of Barrie, ON, with 24% > 100 μg/L for SRP. For both studies, high SRP concentrations were associated with geochemically reducing conditions, suggesting natural aquifer or stream sediment materials may have been a primary source. It is possible that some of these data were influenced by wastewater sources (septic systems or leaky sewers), as well.

The shallow groundwater concentrations of SRP for Wasaga Beach fall within established ranges for general groundwater in both urban and non-urban areas, as discussed above. They are far lower than the peak values noted for wastewater plumes emanating from a specific point source, such as the P plumes at CS-5 and CS-6 (with SRP > 800 μg/L). This provides further support to the conceptual model that the beach groundwater has not (yet) been affected by the potential multitude of P plumes emanating from the hundreds of decommissioned septic systems of the town of Wasaga Beach.

Compared to the open water of Georgian Bay, where SRP concentrations of <1 μg/L are typical (J. Milne, Environment and Climate Change Canada, 2016; personal communication), the shallow groundwater SRP concentrations were high. Thus, groundwater discharging to the shoreline will provide the shallow sediments a steady environment of relatively high dissolved phosphorus, though this may sorb or precipitate onto the shallow sediments with exposure to oxygen along this surface water-groundwater interface (Lewandowski et al. 2015). Of note, the median groundwater SRP concentrations are similar to total phosphorus (TP) concentrations reported for the Nottawasaga River (mean = 40 μg/L, Louis Berger Group and Greenland Int. Inc., 2006).

Groundwater P loading model

Here we calculate a rough (order-of-magnitude) estimate of P loading to Nottawasaga Bay from groundwater discharging along the ~10-km length of urban area of Wasaga Beach. We then extrapolate from this to estimate groundwater P loading from a 50-km length of shoreline that includes Wasaga Beach and many similar sand beaches along the east shore of Nottawasaga Bay (Fig. 1b). The calculation of potential groundwater P loads from presumed natural sources is based on the shoreline surveys' groundwater SRP concentrations and the steady-state groundwater flow information for the CS-5 site, with assumptions that these are broadly representative of conditions of these sand beaches. The available data on which to estimate potential loadings from decommissioned and current septic systems is limited to the 2 identified septic P plumes from provincial park washroom facilities (CS-5, CS-6), which are likely not representative of loads from the more numerous homes and seasonal cottages. Thus, potential maximum septic P loadings are estimated from wastewater production calculations following a protocol similar to that of the LCM (Paterson et al. 2006), which will be described further below. We acknowledge that there is a large amount of uncertainty associated with the following calculations. However, we believe a ball-park figure or range of likely loads is useful to inform an area of importance (i.e., Great Lakes) without any available data on groundwater inputs of P.

For the natural P loading estimate we consider horizontal steady (diffuse, not focused) flow through the top 2 m of the sand aquifer saturated-thickness, as this coincides generally with the depths sampled in the shoreline surveys (from 0.5–1.2 m depth). Also, SRP concentrations in samples from the multi-level wells (Fig. 4) were higher at shallow depths, often with sharply declining concentrations below ~2 m, commonly to ≤10 μg/L. Thus, the suspected smaller SRP mass flux from deeper groundwater is ignored at this point, making this a conservative estimate. Groundwater flux (= v/n) at CS-5 site was calculated to be 38.5 m^3/m^2/a based on the v estimate of 110 m/a and an assumed n of 0.35. Applying this to the 2-m thick top portion of the aquifer and the ~10-km length of beach (Fig. 1) results in groundwater discharge of 770,000 m^3/a. Considering a median SRP concentration of 50 μg/L, which represents the background level not influenced by spikes from septic P plumes and other point sources and which is similar to other reports for natural groundwater, yields a P mass discharge of 40 kg/a from discharging shallow groundwater along this 10 km section of Wasaga Beach.

The estimate of the potential maximum loading from the decommissioned septic systems of Wasaga Beach is based on estimates of P added with wastewater to the town dwellings septic tanks and assuming there is no loss of P in the vadose zone or during transport in the aquifer, as in the LCM. However, here we ignore the LCM limit on the distance from the lake for which a septic system is included in the loadings estimate. Thus, all wastewater P from the entire town is assumed to reach the groundwater and form plumes that eventually discharge to Nottawasaga Bay. In the 1981 census from Statistics Canada (1981), there were 1930 occupied private dwellings in Wasaga Beach, which we assume were all serviced by individual septic systems. The median length of occupancy was 5 months, indicating a predominance of seasonal cottages. A few of these were likely situated along the Nottawasaga River, which might have captured some of these plumes, but we have no data on this, so we assume all contribute to the bay. The LCM calculates the P load from septic systems as the average annual P input per person (estimated as 0.66 kg P/capita/a; Paterson et al. 2006) multiplied by the average annual household occupancy (estimated for seasonal occupancy as 0.69 capita years/a; Paterson et al. 2006), which gives 0.45 kg/a. Thus, the estimated maximum P load to groundwater from the entire town in 1981, the assumed peak in the number of functioning septic systems, is 870 kg/a. However, this load won't reach the bay all at the same time. As illustrated in Fig. 3, this annual load to groundwater will be dispersed over time, from decades to centuries perhaps, due to the range of arrival times of the plume slugs coming from different distances (several km) from the shore and their truncated length resulting from decommissioning. For simplicity, we assume cottage distribution is even across town, covering 2 km distance from shore. Breaking this distance from shore into 50-m sections (rough separation distance between plume arrival overlap), gives a maximum annual P load to the bay from septic systems of 22 kg/a, which is of a similar order-of-magnitude as that from the shallow (top 2 m) background groundwater. With potential P losses due to mineral precipitation in the vadose zone and aquifer, uptake by vegetation (riparian trees), or groundwater capture by drainage ditches, this load could be much less. In addition, due to variability in septic system density, the range of distances from shore, and years of use, this load will vary with the location along the beach and over time, eventually dropping to zero when all septic P plumes are flushed out.

Extrapolation of the estimate of background groundwater P loading for the 10-km urban stretch of Wasaga Beach to the ~50-km total length of sand beaches along the east shore of Nottawasaga Bay results in a P mass loading of 200 kg/a. This requires assuming similar conditions with respect to background P concentrations, shallow groundwater flows to the bay, and the fate and transport of P in the aquifer, as determined or assumed for Wasaga Beach, apply to this larger stretch of shoreline. The strength of this assumption is solely based on the similar geology, topography, and formation of these sand beaches, along with assumed similar recharge to groundwater, leaving much room for uncertainty. The eastern shore of Nottawasaga Bay is lined with cottage communities that still rely on septic systems; however, an estimate of the P load from septic systems for the additional 40-km length of shoreline is difficult to determine due to the lack of compiled historical information on number of dwellings and occupancy. It is clear that the average number of dwellings per length of shoreline today is much less than that of Wasaga Beach (rough estimate <25%); a convenient estimate applied here is that these other beach areas contain the same number of cottages as Wasaga Beach had in 1981 (lower density but greater length). Thus, a rough estimation of the maximum P load from septic systems of the other beaches is 870 kg/a. As for Wasaga Beach, the current P load is likely only from septic systems close to the shore. However, unless these septic systems are decommissioned in favour of community sewer services, the load will increase over subsequent decades to this maximum value as inputs from plumes from further septic systems are added. Additional cottage development could add further to this. When combined with the load from Wasaga Beach (22 kg/a), this gives a maximum total P load from septic systems along this 50-km beach area of nearly 900 kg/a, though the Wasaga Beach load will decline to zero many decades later due to the decommissioning. Again, this calculation assumes similar P fate and transport behaviours apply through the vadose zone and the sand aquifer across this 50-km stretch of sand beach.

For perspective, P mass discharge of the Nottawasaga River, which drains an area of ~3147 km^2 and is the dominant surface water inflow to Nottawasaga Bay, has been estimated as 47,000 kg/a (Louis Berger Group and Greenland Int. Inc., 2006). The calculation of shallow groundwater loading of background P to the 50-km length of beaches is only ~0.4% of that from the river. The calculated complimentary load of septic

P with groundwater would currently add <0.4%, perhaps reaching 2% in several decades. We again emphasize that these loadings should only be considered as approximate, order-of-magnitude estimates. Of most interest is the chance that these are underestimated. The greatest uncertainty likely resides with the groundwater discharge calculation; it is quite conceivable that this could be higher by 3 times or more, considering that the maximum groundwater velocity calculated was 1 m/d (compared to the best guess 0.3 m/d used in these P discharge calculations). The background P concentration seems unlikely to be substantially higher, given its similarity to value ranges from the literature. The addition of P inputs from slightly deeper groundwater (>2 m depth), at likely lower SRP concentrations (Fig. 4), which was ignored in the above calculations, could possibly raise the natural P load by 2–3 times that of the top 2 m of shallow groundwater in our opinion. And yet, even at an upper range, the estimated P load from shallow beach groundwater is still relatively minor (<3% currently, perhaps rising to <5% in several decades) compared to that of the Nottawasaga River.

Conclusions

The discovery of two P plumes emanating from the decommissioned septic beds of provincial park comfort stations (public washroom facilities) proves that septic wastewater can supply P to groundwater in the beach sands of Nottawasaga Bay. This fits the developing conceptual model (Robertson 2003) that P is not extensively trapped as mineral precipitates in the filter sands or soil just below the septic leaching bed in calcareous sediments, thus resulting in groundwater plumes of high dissolved P concentrations. In contrast, such vadose zone trapping appears to be common for non-calcareous sediments. Furthermore, the presence of these P plumes within a few tens of meters from their septic bed, ~15–31 years after decommissioning, demonstrates their longevity. Properties of the plumes suggest they are the remnants of old septic plumes slowed by sorption to the aquifer sediments, rather than being sourced by P flushed from the vadose zone or septic bed after decommissioning. Long delays in transport of groundwater contaminants to surface waters are common. For example, Sanford and Pope (2013) used watershed modeling study of nitrate transport to show several decades will be required to see the full effects of current and future BMPs on water improvements for Chesapeake Bay. Though delay was due to transport through deep porous aquifers, so was a watershed-scale effect – at this scale, decade-long delays may be less surprising. However, here for septic P, decade to century-scale delays are occurring for shallow groundwater flows over short distances (100 s to 1000 s of m only); still surprising despite expectations of P sorption from past research. These findings indicate that decommissioning septic systems is unlikely to lead to immediate reductions in septic P inputs to the bay.

Despite the slow transport, investigation of one of these comfort station P plumes indicates it has reached the bay shoreline. However, rough calculations of P plume transport through the sand aquifer with a plug flow model with linear sorption suggest that the majority of any existing P plumes from residential septic systems, being largely further from shore (>100 m), are unlikely to have reached the shoreline yet. Furthermore, these calculations suggest very few, if any, of the septic P plumes have been flushed completely from the aquifer to the bay, though P mass might have been captured by vegetation or discharge to ditches along the way. Thus, despite decommissioning of the town's septic systems 1–3 decades ago, P loading to Nottawasaga Bay from remnant septic system P plumes may still be increasing.

Samples of shallow groundwater from along the 10-km stretch of Wasaga Beach show that it supplies P to the nearshore of Georgian Bay, with relatively high concentrations (average SRP of 40–80 μg/L, but reaching many 100 s μg/L in places) compared to concentrations in the open bay (<1 μg/L). Similar SRP concentrations were found at a reference site, with no past septic systems. This and other evidence suggests the majority of these samples at average concentrations reflect background groundwater conditions, with P sourced from degrading organic matter, bird droppings, and/or the soil-aquifer minerals. Higher concentrations may be from septic systems or natural sources. The fate of the shallow groundwater P once it reaches the shoreline is unknown because the anoxic, Fe-reducing groundwater will meet oxygenated lake water at the sediment interface or below it, with wave-induced circulation of lake water in the shoreline swash zone. Dissolved phosphorus may then precipitate out or sorb onto oxide minerals (likely iron oxides) that form, as has been reported for arsenic in similar conditions (Lee et al. 2014). This P-rich sediment may then be redistributed around the bay by erosional events.

Estimated P loading estimates for background groundwater (~40 kg/a) and septic P plumes (~20 kg/a) from the 10-km length of Wasaga Beach shoreline suggest that P loading from groundwater to the bay falls within an order of magnitude 10–100 kg/a. If this is extrapolated to the ~50-km length eastern shoreline of Nottawasaga Bay, containing Wasaga Beach and similar sand beaches that are also lined with cottages, but at a lower density than Wasaga Beach, then P load would be <3% currently, perhaps rising to <5% in several decades, of that contributed by the Nottawasaga River. Increasing inputs of P from groundwater have been considered as a possible factor in the proliferation of nuisance algae in the nearshore zone of central eastern Lake Huron (Barton et al. 2013). Here, for Georgian Bay, though these P load estimates are associated with substantial uncertainty, the order-of-magnitude values suggest groundwater is less of a concern for mass balance loading compared to the Nottawasaga River. However, groundwater, with high SRP concentrations discharging to the nearshore, may contribute to localized periphyton and macrophyte growth (Périllon and Hilt 2016) and subsequent eutrophication and algal bloom issues.

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Appendix A. Supplementary data

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