The spatial variability of nitrogen and phosphorus concentration in a sand aquifer influenced by onsite sewage treatment and disposal systems: a case study on St. George Island, Florida

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“Capsule”: Increasing the setback to 50 m and raising drainfields 1 m above groundlevel could reduce nutrient inputs.

Abstract

Groundwater from a shallow freshwater lens on St. George Island, a barrier island located in the Panhandle of Florida, eventually discharges into Apalachicola Bay or the Gulf of Mexico. Nutrient concentrations in groundwaters were monitored downfield from three onsite sewage treatment and disposal systems (OSTDS) on the island. Estimates of natural groundwater nutrient concentrations were obtained from an adjacent uninhabited island. Silicate, which was significantly higher in the imported drinking water relative to the surficial aquifer on St. George Island (12.2±1.9 mg Si l\textsuperscript{-1} and 2.9±0.2 mg Si l\textsuperscript{-1}, respectively), was used as a natural conservative tracer. Our observations showed that nitrogen concentrations were attenuated to a greater extent than that of phosphorus relative to the conservative tracer. At the current setback distance (23 m), both nitrogen and phosphate concentrations are still elevated above natural levels by as much as 2 and 7 times, respectively. Increasing the setback distance to 50 m and raising the drainfields 1 m above the ground surface could reduce nutrient levels to natural concentrations (1.1±0.1 mg N l\textsuperscript{-1}, 0.20±0.02 mg P l\textsuperscript{-1}). © 2002 Elsevier Science Ltd. All rights reserved.

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1. Introduction

Groundwater discharge provides a significant amount of nutrients and contaminants into some coastal zones (Valiela et al., 1978, 1990; Valiela and Teal, 1979; Capone and Bautista, 1985; Lapointe and O’Connell, 1989; Capone and Slater, 1990; Lapointe et al., 1990). In areas with a shallow freshwater system, groundwater may easily be contaminated from onsite sewage treatment and disposal systems (OSTDS), which are typically installed less than 1 m above the water table and may be flooded during heavy rains. The transport of non-point source pollution from OSTDS to shallow groundwater and ultimate discharge to surface waters could be an important source of contamination to the marine environment, especially in areas of restricted circulation such as an estuary or small embayment. While this pathway is becoming better known, it is still largely ignored by scientists and coastal managers alike. When evaluating impacts from OSTDS on coastal waters, it is important to consider the natural flux of nutrients in order to implement sound management strategies.

St. George Island, like many barrier islands, forms the outer perimeter of an estuary (Apalachicola Bay) and is critical to the bay’s productivity because its orientation determines the salinity distribution as well as other water quality features of the bay. Apalachicola Bay is one of the most economically important estuarine systems in Florida due to oyster and shrimp harvesting. The island, which is located within the Apalachicola National Estuarine Reserve, is developing at a rapid pace. This has resulted in a relatively high density of OSTDS on the island. Growth in these and surrounding...
communities is of major concern with regard to the health of the estuary, which is likely sensitive to slight physical, chemical, and biological perturbations. Although barrier islands play a critical role in this balance, very little is known about the groundwater dynamics and potential impacts of contaminated groundwaters on the surrounding waters.

In order to prevent the possible deterioration of Apalachicola Bay and other estuarine systems, including economic zones (oyster beds and areas of dense shrimp populations), contaminants of any type must be monitored closely. Although the Apalachicola River (the largest river in Florida) clearly provides the majority of the nutrients to the bay, those supplied via groundwater transport from St. George Island may be locally important. Without knowledge of the groundwater contribution, interpretation and management decisions concerning the treatment of sewage may be faulty and lead to future environmental threats. Thus, monitoring of OSTDS in an area of increasing development and density is necessary to help guide future wastewater treatment decisions.

Virtually all the homes and businesses on St. George Island have an onsite wastewater treatment and disposal system with a drainfield less than 1 m above the shallow water table. We showed earlier using artificial tracers (Corbett et al., 2000) that average groundwater velocities are as high as 0.4 m day\(^{-1}\) and presumably are even greater during large rain events. The weakly dispersive nature of the island’s aquifer and the fairly rapid transport rates may make current minimum distance-to-surface water (~23 m) regulations for permitting wastewater systems inadequate for protecting local surface water quality. We collected wastewater as it exited the OSTDS and monitored the groundwater nutrient concentrations in the surrounding groundwaters at selected sites on St. George Island (Fig. 1). Multi-level samplers (MLS) and 5 cm PVC monitoring wells were placed downgradient from three wastewater systems at different locations on the island. These sites included two types of onsite sewage treatment and disposal systems, aerobic and anaerobic (septic) treatment. Compositional differences in the effluent discharged and possible groundwater contaminants from both types of systems have been evaluated. The wells were monitored over the course of the study for nutrient concentrations along the flow path toward surface waters. Two sampling locations on undeveloped Little St. George Island (Fig. 1), a small barrier island adjacent to the main island was established to assess reasonable background levels.

2. Study site

2.1. St. George Island

St. George Island, a barrier island in the Panhandle of Florida, is approximately 48 km long and averages less
than 0.5 km in width. Dr. Julian G. Bruce State Park occupies the east end of the island. The climate in the region is mild with a mean annual temperature of approximately 20°C (Livingston, 1984). The mean annual rainfall over the area, recorded over the last 42 years by the NOAA weather station in Apalachicola, is approximately 140 cm, 40% occurring during the late summer months. The tidal range in the bay and Gulf is less than 0.5 m. The surficial aquifer on the island is composed of medium to fine sand grains overlying a silty clay impermeable barrier between 7.6 and 9.2 m below the surface that forms a base to the aquifer (Livingston, 1984). Water in the shallow freshwater lens is primarily derived from rainfall and eventually discharges into Apalachicola Bay or the Gulf of Mexico. The impermeable clay layer separates rain-derived freshwater from the surrounding salt water. St. George Island has a characteristically shallow water table, which increases the probability of OSTDS-derived groundwater contamination and transport to the surrounding marine waters.

2.2. Experimental sites

Experimental sites were selected according to location on the island, proximity to Apalachicola Bay, type of OSTDS, and the amount of time the residence was occupied. Care was given to locate sites adjacent to the bay with a similar beachfront, i.e. no canals, natural topographic gradients, etc. Three sites were chosen, including a site located within the Dr. Julian G. Bruce State Park (SP Site) on the far eastern end of the island, a private residence near Bob Sikes Cut (BL Site) on the extreme western end of the island, and another private residence approximately 3 km west of the causeway (JA Site) between the other two locations (Fig. 2). Each site had an average of three residents and was occupied year round. Detailed descriptions of the sites have been given earlier (Corbett et al., 2000). Briefly, the SP Site is within the state park and consists of 12 5-cm monitoring wells and thirteen multi-level samplers. The OSTDS is a septic tank located approximately 100 m from the bay. Nutrient samples were collected on a monthly basis from September 1997 to May 1999. The JA Site consists of seven monitoring wells and seven multi-level samplers. The OSTDS is an aerobic system approximately 60 m from the bay. The BL Site consists of seven monitoring wells and eight multi-level samplers. The OSTDS is a septic system raised 1 m above ground level and is approximately 70 m from the bay. Samples for nutrients were collected at the JA and BL Sites monthly from April 1998 through March 1999. Water level heights were measured in the wells monthly in order to evaluate the piezometric surface, hydraulic gradients, and groundwater flow direction at each site. Details of the hydrogeologic studies, including flow rate determinations, have already been reported in Corbett et al. (2000). Groundwater flow was typically toward the bay with dramatic changes in hydraulic gradients (<0.001–0.008) associated
with rain events. In addition, several artificial tracer tests were completed to gather additional information on the island hydrogeology. The hydraulic conductivity of the island was estimated to be 36 m day\(^{-1}\) with groundwater horizontal transport rates between 0.02 and 0.42 m day\(^{-1}\), dependent on site and season (Corbett et al., 2000).

2.3. Background site

Little St. George Island, also referred to as Cape St. George, was once part of St. George Island. In 1954, the Corps of Engineers separated Cape St. George from the larger island in order to build a navigable channel to the Gulf, now called Bob Sikes Cut. In 1977, the State of Florida purchased Little St. George Island under the Environmentally Endangered Lands Program and created a state preserve. Due to the island’s status, there has been no development, thus the groundwater nutrient concentrations have had no influence from wastewater systems. Two multi-level samplers were installed on the island, one within 1 km of the east end of the island and the other approximately 5 km west of the east end. The stratigraphies at these well locations are almost identical to the experimental sites. Concentrations of nutrients from these background well sites are considered representative of natural background levels.

3. Materials and methods

Monthly measurements of hydrographic and chemical variables were made from September 1997 to May 1999 at the SP site and from April 1998 to March 1999 at the BL and JA field sites. Water samples were collected in 125 ml acid washed polypropylene bottles. Samples for nutrient analyses were filtered through GF/F filters (0.7 µm) in the field and were stored on ice in the dark until analysis. Nutrients were analyzed within 24 h of sample collection. Nitrate (NO\(_3^-\)) and total dissolved nitrogen (TN) concentrations were determined with the chemiluminescence detector-based method for trace nitrate (NO\(_3^-\)) and NO\(_3^-\) in filtered aqueous samples. Total nitrogen samples were first digested at 120°C for 30 min in the presence of persulfate (Pujo-Pay and Raimbault, 1994). The chemiluminescence method was developed by Cox (1980) and applied to seawater analyses by Garside (1982). We used the modified version of Garside’s method (Braman and Hendrix, 1989). Nitrate and NO\(_3^-\) are quickly reduced to nitric oxide in an acidic medium containing vanadium (III) at 80–90°C. Nitric oxide is then removed from the reaction solution by scrubbing with helium carrier gas and is detected with a Thermo Environmental Model 42 chemiluminescence NOx analyzer connected to a HP 3396 Series II integrator. Nitrite concentrations were measured colorimetrically (Stickland and Parsons, 1972) and subtracted from the NO\(_3^-\) + NO\(_2^-\) values to yield NO\(_3^-\) concentrations. Ammonium (NH\(_4^+\)) was determined with the phenol-hypochlorite method as described by Strickland and Parsons (1972). Soluble reactive phosphate (PO\(_4^{3-}\)) was determined by the ascorbic acid-phospho-molybdate method outlined in Strickland and Parsons (1972). Total dissolved phosphate (TP) was put through the same procedure following a digestion at 120°C for 30 min in the presence of persulfate. Silicate was determined using the molybdate procedure described in Strickland and Parsons (1972).

4. Results and discussion

4.1. Nutrient dynamics

St. George Island’s drinking water is obtained from three deep wells (>150 m) on the mainland, which is then pumped over to a large storage tank (570 m\(^3\)) on the island. Interestingly, the silicate concentration in the tap water averaged 11.5 mg Si l\(^{-1}\), ranging from 8.1 to 14.3 mg Si l\(^{-1}\) throughout the study period at all the experimental sites, which is four times higher than the concentration in the unimpacted surficial aquifer. Groundwater collected from Little St. George Island, indicative of natural concentrations, had an average silicate concentration of 2.9 mg Si l\(^{-1}\), ranging from 1.1 to 3.9 mg Si l\(^{-1}\). The reactivity of silicate in this type of environment, a surficial aquifer of primarily medium to fine sand, should be slight and the concentrations measured are at or near the saturation value for pure quartz (approximately 2.8 mg Si l\(^{-1}\)), and well below saturation with other silica phases. The kinetics for dissolved silica precipitating to low-temperature quartz are so slow that it would take millions of years, much longer than the residence time of the groundwater. Therefore, we evaluated the prospect of using silicate as a conservative indicator of the wastewater plume in this study. Routine sampling of the monitoring network demonstrated that the effluent plume could be easily distinguished in the field by measurement of the groundwater silicate concentration (Fig. 3). Using silicate as an indicator shows the wastewater plume at the SP site (January 1998 data) extending almost to surface waters with relatively little dilution. Artificial tracer experiments from previous work indicated this same path and approximate dilution for the wastewater plume (Corbett et al., 2000). Therefore, silicate serves as a natural tracer of the wastewater due to the origin of the potable water for the island.

Nutrient concentrations in the subsurface varied considerably, with highest concentrations near the OSTDS typically occurring during periods of increased rainfall. However, general patterns in the data were observed and these simplified trends were used for interpretation.
For instance, the majority of the nitrogen species measured were either in the form of ammonia or organic nitrogen. Nitrate and nitrite represented less than 1.5% on average of the total amount of nitrogen in all the samples collected at the experimental sites. On the other hand, soluble-reactive phosphate averaged between 85 and 90% of the total phosphate measured. Based on these observations, we will present trends for total nitrogen, ammonia, total phosphate, and silicate. Therefore, unless otherwise indicated, we assume that nitrate and nitrite are negligible and soluble-reactive phosphate is approximately equal to total phosphate.

Nutrient concentrations of wastewater effluent collected just before entering the drainfield at each site are extremely elevated above natural levels (Table 1) and similar to values reported elsewhere (Harris, 1995; Harmon et al., 1996; Ptacek, 1998; Robertson and Harman, 1999). Although elevated concentrations were initially placed into the subsurface, both nitrogen and phosphate concentrations decreased as the plume moved away from the drainfield (Table 2). Both nitrogen and phosphate showed immediate reductions as much as an order of magnitude in the closest well sampled, with very little reduction in silicate concentrations. Concentrations of total nitrogen in the groundwater was reduced by an additional 70% at the SP site and 40% at the JA site as the plume moved toward surface waters. The nitrogen concentrations at these two sites approach natural levels after approximately 50 m travel away from the drainfield. Interestingly, the nitrogen concentrations at the BL site are not significantly different ($P<0.01$) from natural concentrations throughout the well field. In fact, wells closest to the drainfield (within 5 m) have nitrogen concentrations equal to natural concentrations. We attribute this 95% reduction in nitrogen levels to the elevated drainfield (see discussion in Section 4.3). However, phosphate concentrations at all three experimental sites are still significantly elevated (0.5–0.7 mg P l$^{-1}$) above the natural level (0.2 mg P l$^{-1}$), although significantly lower than tank effluent (3.4–8.1 mg P l$^{-1}$), at wells beyond 50 m from the drainfields. As with the nitrogen levels, the largest reduction in phosphate concentration was observed at the BL site. The observed reductions in nutrient concentrations as the
plume moves through the aquifer can be attributed to dilution, biogeochemical reactions (denitrification, biological uptake), sorption, and/or precipitation.

We have shown that silicate appears to act conservatively in this environment and is significantly higher in the wastewater than the natural aquifer (Table 1). Therefore, normalizing the nutrient concentrations to the amount of silicate at each well will account for any dilution that has occurred; highlighting changes in nutrient concentrations associated with non-conservative behavior as the plume moves downgradient (Fig. 4). It is apparent that nitrogen and phosphate are initially removed relative to the tank effluent (0 m) at each site. Downfield reductions in nutrient concentration by dilution and other non-conservative processes are evident at each site except the BL site (Table 2; Fig. 4). At the SP site, there appears to be very little change in the normalized nutrient concentrations after a distance from the drainfield of approximately 35–50 m. Normalized phosphate concentrations at the JA site continue to show reductions downgradient throughout the site. However, there is little difference between the normalized nutrient concentration and distance from the drainfield at the BL site, possible due to the design of the OSTDS (Section 4.3.).

### 4.2. Nutrient fluxes to surface waters

Annual groundwater nutrient contributions to Apalachicola Bay were calculated using estimates of groundwater flow together with the groundwater nutrient concentrations at the point of discharge. Corbett et

<table>
<thead>
<tr>
<th>Site</th>
<th>Well No.</th>
<th>Distance from drainfield (m)</th>
<th>Total nitrogen mg N l⁻¹</th>
<th>Ammonium mg N l⁻¹</th>
<th>Total phosphate mg P l⁻¹</th>
<th>Silicate mg Si l⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>SP site</td>
<td>14 (64)</td>
<td>14</td>
<td>3.1±0.7ᵃ</td>
<td>1.0±0.3</td>
<td>1.2±0.1ᵃ</td>
<td>5.8±0.3ᵃ</td>
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<td></td>
<td>15 (59)</td>
<td>26</td>
<td>2.4±0.8ᵃ</td>
<td>0.6±0.2ᵃ</td>
<td>0.8±0.1ᵃ</td>
<td>5.7±0.3ᵃ</td>
</tr>
<tr>
<td></td>
<td>17 (35)</td>
<td>35</td>
<td>1.0±0.1</td>
<td>0.2±0.1</td>
<td>0.6±0.1ᵃ</td>
<td>6.7±0.7ᵇ</td>
</tr>
<tr>
<td></td>
<td>7 (21)</td>
<td>54</td>
<td>1.3±0.2</td>
<td>0.4±0.1</td>
<td>0.5±0.0ᵇ</td>
<td>5.3±0.1ᵇ</td>
</tr>
<tr>
<td></td>
<td>8 (9)</td>
<td>63</td>
<td>0.8±0.1</td>
<td>0.2±0.0</td>
<td>0.6±0.0ᵇ</td>
<td>6.3±0.4ᵇ</td>
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<td></td>
<td>9 (21)</td>
<td>86</td>
<td>1.0±0.1</td>
<td>0.4±0.0</td>
<td>0.6±0.0ᵇ</td>
<td>4.9±0.3ᵇ</td>
</tr>
<tr>
<td>JA site</td>
<td>4 (37)</td>
<td>7</td>
<td>3.2±0.6ᵃ</td>
<td>1.8±0.4ᵃ</td>
<td>1.5±0.2ᵃ</td>
<td>10±1ᵃᵇ</td>
</tr>
<tr>
<td></td>
<td>7 (12)</td>
<td>25</td>
<td>1.9±0.1ᵃ</td>
<td>1.0±0.1ᵃ</td>
<td>1.4±0.2ᵃ</td>
<td>9.2±0.7ᵇ</td>
</tr>
<tr>
<td></td>
<td>8 (34)</td>
<td>37</td>
<td>2.2±0.2ᵃ</td>
<td>1.2±0.2ᵃ</td>
<td>0.9±0.1ᵇ</td>
<td>8.4±0.3ᵇ</td>
</tr>
<tr>
<td></td>
<td>9 (12)</td>
<td>50</td>
<td>1.7±0.6</td>
<td>0.8±0.4</td>
<td>0.7±0.1ᵃ</td>
<td>8.5±0.5ᵇ</td>
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<tr>
<td>BL site</td>
<td>3 (29)</td>
<td>5</td>
<td>1.0±0.1</td>
<td>0.6±0.1ᵃ</td>
<td>0.2±0.0</td>
<td>4.4±0.3ᵇ</td>
</tr>
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<td></td>
<td>4,5 (38)</td>
<td>9</td>
<td>0.8±0.1</td>
<td>0.4±0.1</td>
<td>0.8±0.2ᵃ</td>
<td>7.7±0.9ᵇ</td>
</tr>
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<td>6 (2)</td>
<td>21</td>
<td>0.6±0.2</td>
<td>0.6±0.5</td>
<td>0.3±0.1</td>
<td>7.3±0.8ᵃ</td>
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<tr>
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<td>7,8 (18)</td>
<td>25</td>
<td>1.1±0.1</td>
<td>0.7±0.1ᵃ</td>
<td>0.3±0.1</td>
<td>6.7±1.3ᵃ</td>
</tr>
<tr>
<td></td>
<td>9 (20)</td>
<td>37</td>
<td>0.9±0.1</td>
<td>0.3±0.1</td>
<td>0.5±0.1ᵃ</td>
<td>5.5±0.3ᵇ</td>
</tr>
<tr>
<td></td>
<td>13,14 (35)</td>
<td>57</td>
<td>1.0±0.1</td>
<td>0.6±0.1ᵃ</td>
<td>0.3±0.1</td>
<td>5.5±0.6ᵇ</td>
</tr>
</tbody>
</table>

ᵃ Concentrations are significantly \( P < 0.05 \) higher than the natural aquifer.
ᵇ Concentrations are not significantly \( P > 0.1 \) different than the effluent nutrient concentrations.
al. (2000) estimated the total amount of groundwater discharging into the bay from the island to be between 1–9×10^6 m^3 year^{-1}. This discharge is a range estimated by two independent techniques: (1) a water balance approach which accounted for sources and sinks to the surficial aquifer, assuming 50% of the water discharges into the bay; and (2) groundwater velocity measurements obtained through tracer experiments at two locations on the island and an estimate of the aquifer thickness and length. Both techniques gave a range in discharge that agreed very well.

Estimating the groundwater nutrient concentration at the point of discharge is typically difficult due to chemical and biological alterations during transit. It is apparent that the nutrient concentrations are significantly reduced as surface waters are approached (Figs. 4–5). As an upper limit of total nitrogen and phosphate concentrations discharging to surface waters, we used the average of the well closest to the surface waters at the JA site (well No. 9, Table 2). Average concentrations at the other two sites were similar but slightly lower. As a lower limit, we measured nutrient concentrations of interstitial waters directly offshore at all three experimental sites using a “peeper,” a close interval porewater sampling device (Hesslein, 1976).

The lowest interstitial total nitrogen concentration was 0.4 mg N l^{-1}, observed at the BL site. The lowest interstitial total phosphate concentration was 0.3 mg P l^{-1}, observed at the JA site. A summary of the data used to estimate the nutrient fluxes to the bay is presented in Table 3.

The total nitrogen flux into Apalachicola Bay ranged between 1.5 and 65.7 mg N m^{-2} year^{-1}, accounting for the entire area of the bay, while the total phosphate flux ranged between 1.2 and 24.2 mg P m^{-2} year^{-1}. These nutrient fluxes are a minor fraction of that supplied by the Apalachicola River which is not surprising considering the size of the river and associated drainage basin. In addition, the nutrient concentrations in the groundwater at the point of discharge at all the experimental sites were approximately equal to or lower than the concentrations at the background sites. Although this is a fortunate set of circumstances, it should be noted that all of these OSTDS were set back significantly further from surface waters than current regulations permit. Also, many bar-built estuaries do not necessarily have a large river dominating the nutrient supply to the system. Ecosystems without a significant source of direct discharge would likely be more susceptible to alterations in the groundwater input and associated contaminants.

4.3. Management issues

The State of Florida requires all wastewater systems to be setback at least 23 m (75 ft) from the mean high tide water line. In addition, since 1992 all wastewater systems installed or replaced on St. George Island are required by the county to be aerobic treatment units, since aerobic wastewater treatment systems provide a higher level of treatment relative to septic systems (National Small Flows Clearinghouse, 1996). The data presented here also indicates that the aerobic tank is more efficient in removing nutrients than septic systems (Table 1). Assuming the three households had similar volumes of waste (occupancy was typically between three and four residents), the difference in effluent exiting the

Table 3

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Minimum</th>
<th>Maximum</th>
</tr>
</thead>
<tbody>
<tr>
<td>Groundwater flux (m^3 year^{-1})^a</td>
<td>1×10^6</td>
<td>9×10^6</td>
</tr>
<tr>
<td>Total nitrogen concentration (mg N l^{-1})</td>
<td>0.4</td>
<td>1.9</td>
</tr>
<tr>
<td>Total phosphate concentration (mg P l^{-1})</td>
<td>0.3</td>
<td>0.7</td>
</tr>
<tr>
<td>Total nitrogen flux^b (mg m^{-2} year^{-1})</td>
<td>1.5</td>
<td>65.7</td>
</tr>
<tr>
<td>Total phosphate flux^b (mg m^{-2} year^{-1})</td>
<td>1.2</td>
<td>24.2</td>
</tr>
</tbody>
</table>

^a Corbett et al. (2000)

^b Flux is based on the entire area of Apalachicola Bay (260 km^2)
tank should give some indication on the level of treatment. Total nitrogen and phosphate concentrations were reduced in the aerobic tank by as much as 50% relative to the two septic tanks. Based on these data it appears the aerobic tank is more efficient at reducing the nutrient load to the drainfield. However, reducing the load to the groundwater and surface waters should be the ultimate goal of any management decision regarding OSTDS.

When evaluating the current setback distance, we felt it important to consider the natural nutrient concentrations in the groundwater, since it should be necessary to remove nutrients to this level to prevent potential impacts to surface waters. Silicate concentrations at the experimental sites are typically two–three times greater than those observed at the background locations (Fig. 5). An elevated silicate concentration indicates influence from wastewater throughout the downgradient monitoring wells. However, as we indicated earlier, nitrogen and phosphorus concentrations were attenuated downgradient relative to the silicate concentrations at the SP and JA sites and nitrogen concentrations at the BL site were at natural levels throughout the site (Figs. 4 and 5). Although nitrogen and phosphorus concentrations are reduced at the current setback distance (23 m) relative to the nearfield wells, they are still well above natural levels. By extending the setback distance to approximately 50 m from surface waters, an additional 50% reduction in nitrogen and phosphorus may be achieved. At this distance, nitrogen concentrations at each sight were not significantly \( P > 0.1 \) different than the natural levels. Phosphate concentrations, although significantly reduced relative to the 23 m setback distance, are still above the natural levels at 50 m from the drainfield at the SP and JA sites. At the BL site, phosphate concentrations were not significantly different than natural levels.

The vertical distance the OSTDS lies above the water table is a key component in minimizing potential impacts from nutrients and pathogens on groundwater and surface waters (Harris, 1995). This is especially true in environments similar to St. George Island, where the water table lies within 0.25 m of the ground surface at many locations. Based on the results reported here, raising the OSTDS above the ground surface can make a significant difference in the surrounding nutrient concentration. The septic system and drainfield at the BL site is raised approximately 1 m above the natural ground surface, resulting in a longer residence time for the effluent in the vadose zone and preventing inundation of the system during heavy rain events. In a typical septic system, the soil in and directly beneath the drainfield is used as the final treatment step, acting as a chemical and biological filter. As the wastewater percolates to the groundwater below, the filtration process and organisms in the soil work together to remove toxins, bacteria, viruses, nutrients, and other pollutants from the wastewater. Raising the drainfield above the natural level by as little as 1 m at the BL site increased the removal of nitrogen and phosphate by as much as 40% compared to the other two experimental sites. The concentrations of total nitrogen at the nearfield wells at the other two sites were very similar to each other and approximately two–three times greater than the observed concentrations at the BL site. The reduction of nitrogen in the drainfield may be attributed to loss by denitrification or perhaps by bacterial assimilation. The mechanism for the nitrogen loss cannot be clearly defined using the available data. In addition, the BL site was the only one of the three that reduced the phosphate concentrations to natural levels within approximately 50 m. The greatest removal of phosphate occurred within the vadose zone of the drainfield and was probably associated with a direct precipitation or adsorption onto particles.

At our experimental sites, had the OSTDS only been setback from surface waters 23 m, the regulated distance, a significant amount of nitrogen and phosphorus would be discharged into surface waters at concentrations well above natural levels. Although potential impacts were not assessed, increasing the current setback distance to approximately 50 m and raising the OSTDS, including the drainfield, 1 m above the ground surface would provide enough of a buffer zone in the system to reduce the nutrients to background concentrations. However, as population density increases, thus increasing OSTDS density, this proposed 50 m setback distance may not be suitable and should therefore be readdressed in the future. A more efficient method for establishing management regulations in changing communities such as these may be long-term monitoring at select locations.

5. Summary

Nutrient concentrations monitored downgradient from wastewater disposal systems show significant attenuation before discharge into surface waters. Silicate was used as a natural conservative tracer, providing insight to the extent of the wastewater plume and dilution of other nutrients. Total nitrogen, ammonia, and total phosphate were all attenuated relative to silicate, indicating non-conservative removal of these nutrients in the subsurface. Estimates of the total nitrogen flux into Apalachicola Bay from groundwater originating on St. George Island ranged between 1.5 and 65.7 mg N m\(^{-2}\) year\(^{-1}\) and the total phosphate flux ranged between 1.2 and 24.2 mg P m\(^{-2}\) year\(^{-1}\), based on estimated groundwater discharge rates and groundwater and porewater nutrient concentrations measured near surface...
waters. However, current setback distances (23 m) are insufficient to reduce nutrient concentrations to natural levels. By increasing the setback distance to approximately 50 m, most nutrient concentrations will be reduced to natural levels.

Nutrient results indicate that the most efficient OSTDS on the island would be an aerobic system that was raised above the natural land elevation by approximately 1 m. There is a significant difference in the tank effluent of the aerobic tank relative to the two septic systems. Raising the OSTDS, such as at the BL site, provides additional time and material to filter the wastewater before it reaches the water table, thus significantly reducing the nutrient load to groundwaters and ultimately surface waters. Therefore, developing an aerobic system that is raised by as little as 1 m from the natural ground surface on St. George Island would likely provide the most efficient means for removing contaminants from the wastewater before it is introduced to the groundwater. In addition, increasing the setback distance to 50 meters from surface waters would provide better protection of surface waters at the current OSTDS density.

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