Groundwater nitrate contamination remediation through permeable reactive barriers on the Forge River, Long Island, New York

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Abstract of the Thesis

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The Forge River on the south shore of Long Island is contaminated by nitrate. The Forge River leads into Moriches Bay. Nitrate contamination in the river leads to algal blooms, hypoxia of the river and eventually fish kills. The area surrounding the Forge River is largely residential and not sewered with the residents relying on septic systems for sewage disposal. Nitrate from even small septic systems has been found in groundwater as far as 100m down gradient.

Other investigators have shown the occurrence of submarine groundwater discharge on Wills Creek, a tributary of the Forge River and my study site. I hypothesized that much of the nitrate contamination was flowing into the river from the surrounding groundwater. In order to remediate the groundwater before it reached the river, the study looked at whether this location would be a suitable location for a permeable reactive barrier. Monitoring wells were installed in April and June and sampled over the months of June and July of 2014. Groundwater analysis included nitrate-nitrite, ammonium, total nitrogen and total organic carbon.

Dissolved oxygen concentrations were below 4.08 mg/L in all of the installed wells. Conductivity was less than 1.37 mS/cm in each well proving that the wells were in fresh groundwater. Nitrate-nitrite ranged from 0.46 mg/L to 22.95 mg/L. Samples with lower dissolved oxygen concentrations seemed to have a higher nitrate-nitrite concentration. Ammonium concentrations ranged up to 19.56 mg/L. Ammonium was highest in the monitoring well that also had the highest nitrate-nitrite concentrations.

Evidence was found for three potential contaminant plumes. Two were at the water-table apparently leading out from the individual septic systems and one was deeper and apparently more widespread. The hydraulic conductivity of a permeable reactive barrier would need to be over 8.2 m/d congruent with the Upper Glacial Aquifer. I concluded that a permeable reactive barrier could work to denitrify groundwater at the Wills Creek study site, provided that it was engineered to allow the correct residence time to occur for denitrification.

Dedication Page

This thesis is dedicated to my husband, Seth Pritt, who supported me during graduate school.

And to my sister, Courtney Anne Minor, who read this thesis more times than scientists previously thought possible.

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Introduction

Nitrate contamination of groundwater is a serious health concern (Robertson et al. 2000). Almost all of Long Island, New York uses groundwater as drinking water, and in Suffolk County, all residential drinking water comes from groundwater. The background level of groundwater nitrate in the United States is set at 3 mg/L NO_3-N (Kolpin et al. 1998). However, many samples collected throughout the United States do not have detectable levels of NO_3, and this number may be too conservative (Spalding and Exner 1993). Nitrate levels in groundwater are a direct result of nitrate levels in the soil (Bielek 1998). In areas where there is not an abundance of agriculture and fertilizer use, nitrate levels have been found lower than where there is fertilization occurring (Oenema et al. 1998).

Drinking water with high nitrate levels can be toxic to infants resulting in a blood disorder known as blue-baby syndrome, or methemoglobinemia (Munster et al. 2010; Safe Drinking Water Committee 1977). The Long Island "208 Study" identified the use of fertilizers and in-ground sewage disposal systems as primary sources of nitrogen into the groundwater (Koppelman 1978). Recently, the draft Suffolk County Comprehensive Water Resources Management Plan (Suffolk County Government 2014) recognized elevated concentrations of nitrate in groundwater. In coastal waters, groundwater sources of nitrogen have been linked with harmful algal blooms along with a loss of eel-grass and salt-marsh habitat (Suffolk County Government 2014). The Forge River on the south shore of Long Island has a longstanding history of nitrate contamination from both surface runoff and groundwater discharge. From the spring to the fall, hypoxia due to algal blooms occurs in the Forge River and causes both ecological and aesthetic problem for the surrounding area (Swanson et al. 2009a). These algal blooms occur due to the large amount of nitrate entering the surface water system.

This project considered the use for permeable reactive barriers (Robertson and Cherry, 1995) as a remediation treatment at the shoreline of Wills Creek, a tributary of the Forge River Figure 1).

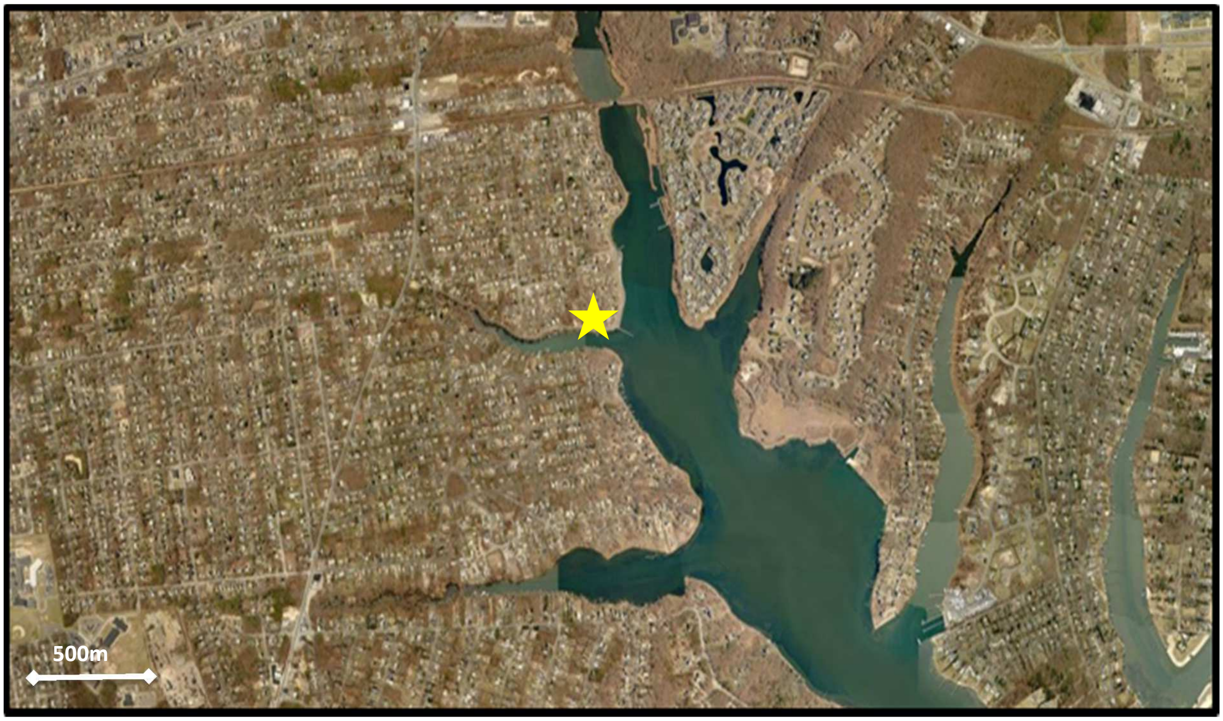

Figure 1. Location of the research site. Yellow star marks research site located on Riviera Drive on the southern shore of Long Island, Mastic Beach, Town of Brookhaven, New York. (Picture from Google Earth 2014)

Permeable Reactive Barriers

In order to remove nitrate from groundwater, the nitrate must undergo denitrification. Initially, microbiologically mediated reactions are used to transform nitrogenous compounds to a form that can be passed readily through aquifers (Soares, 2000). Denitrification is the microbially mediated process which reduces NO_3^- to N_2O or N_2 (Korom, 1992). There are four requirements for denitrification:

(1) N oxides (NO_3^-, NO_2^- and N_2O) as terminal electron acceptors.

(2) Metabolically capable bacteria present.

(3) Suitable electron donors.

(4) Anaerobic Conditions.

In groundwater, biologically mediated denitrification can occur naturally as bacteria use nitrate as an electron acceptor in the absence of oxygen (Soares, 2000). This process turns nitrate into nitrogen gas (N_2) after some intermediary transformations, eventually removing it from the groundwater system.

Heterotrophic denitrifying bacteria use nitrate as a terminal electron acceptor under suboxic and anoxic conditions (Libes, 2009). Biological denitrification of nitrate consists of a sequence of enzymatic reactions where nitrate is reduced to nitrogen gas through a series of inorganic nitrogen intermediates (Soares, 2000):

$$NO_3^- \rightarrow NO_2^- \rightarrow NO + N_2O \rightarrow N_2$$

The denitrification process ends with molecular nitrogen because there are no nitrogen fixers in groundwater. Once the molecular nitrogen reaches saturation in the groundwater, it migrates out of the saturation zone (Korom, 1992). If oxygen levels are high, the denitrifying bacteria will switch from using nitrate to more energetically favorable oxygen as the electron acceptor, and N_2 gas will no longer be created.

Most denitrifying bacteria are capable of surviving with or without oxygen because they are heterotrophic facultative anaerobes (Payne 1981). These bacteria are present throughout the sedimentary column. When oxygen is present, it represents the electron acceptor with the most energy (Burdige 2006). In groundwater systems with anoxic conditions the only limiting factor is the concentration of electron donors (Korom 1992). If the concentration of electron donors is the limiting factor, biological denitrification treatment usually consists of providing suitable carbon and energy sources (Soares 2000).

A carbon source is necessary in the groundwater for denitrification for two reasons. First, the carbon source acts as an electron donor. This means in this reduction-oxidation reaction it is the part being oxidized (Libes 2009). Carbon is also needed as a substrate for the denitrifying bacteria. The bacteria use this reaction to increase their biomass. If there was no biomass increase, there would be no need for the bacteria to denitrify.

Traditionally, the methods for removing nitrate contamination from groundwater are invasive to the ecosystem and technically complex. Many involve injecting a chemical solution into the groundwater or pumping the contaminated water out of the ground on a semi-regular basis (Gavaskar 1999). Recently however, techniques for removing nitrate contamination associated with septic systems have moved towards passive, mechanically simple methods that typically do not require maintenance after installation (Robertson and Cherry 1995). The development of these permeable reactive barriers (PRBs) may present an alternative to more expensive and invasive traditional methods of groundwater remediation (Gupta and Fox 1999).

PRBs are constructed by placing a porous wall perpendicular to the flow of groundwater (Schipper et al. 2004). PRBs contain waste organic matter, such as sawdust or wheat grass. This provides a carbon source for aerobic bacteria to utilize the oxygen and create an anoxic environment which promotes denitrification (Schipper et al. 2010). PRBs have been shown in multiple studies to reduce nitrate levels in groundwater for many different periods of time (Schipper and Vojvodic-Vukovic 2001; Robertson et al., 2008; Slater and Binley 2003).

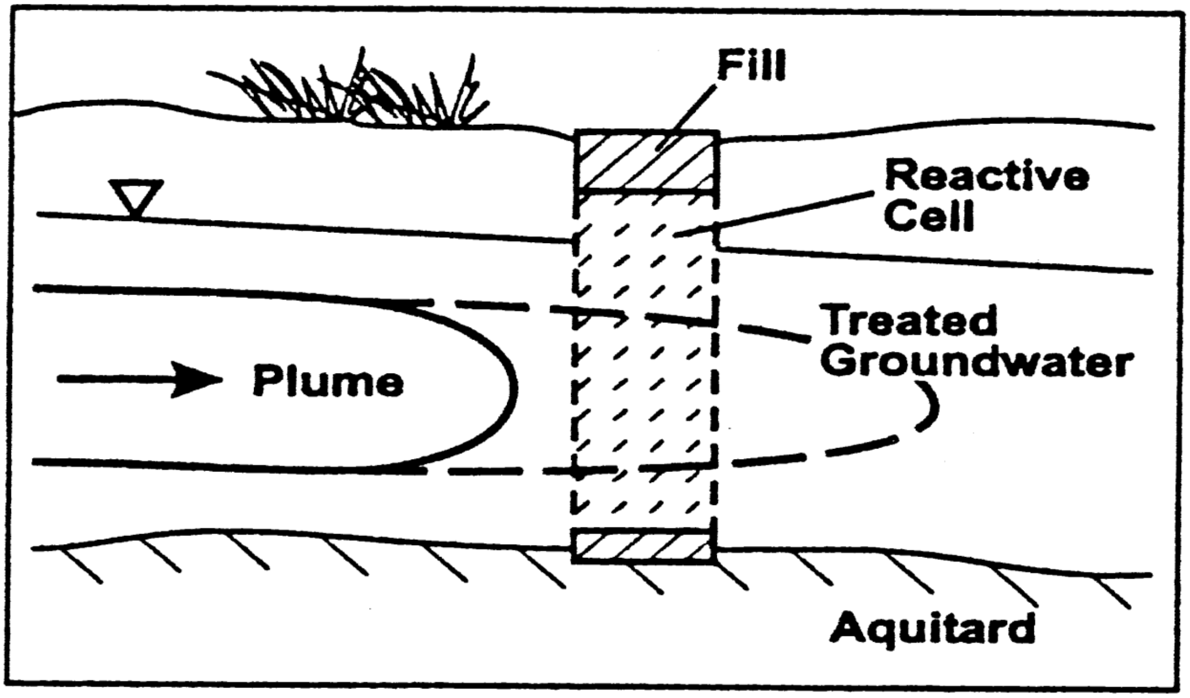

Figure 2. Installed Permeable Reactive Barrier. The groundwater with the contamination plume is treated to remove the contaminant as it flows through the reactive cell (from Gavaskar 1999).

PRBs are installed subsurface at the level of the water table, typically as a wall through which horizontal groundwater flow will travel (Figure 2). The high startup cost and the large construction area due to the trenching are initial issues for the use of PRBs. However, PRBs are considered more cost effective than injection treatments or pump-and-treat methods (Robertson et al. 2008; Robertson et al. 2005).

Permeable Reactive Barriers (PRBs) to reduce nitrogen loading into the coastal zone was looked into by the National Oceanographic and Atmospheric Association (NOAA) in conjunction with the University of New Hampshire's Institute for Coastal and Estuarine Environmental Technology (CICEET). In this study two pilot-scale PRBs were installed along the banks of Waquoit Bay and Childs River in Cape Cod, MA (Vallino et al. 2008). These PRBs were approximately 10-20 m long and filled with NITREX™ medium (a mixture of woodchips and lime). The PRBs were sampled several times during the course of the CICEET project. The results showed that the nitrate entering the PRBs at both sites was effectively removed from the system. With the nitrate removed there was a drop in algal production in the area (Figure 3).

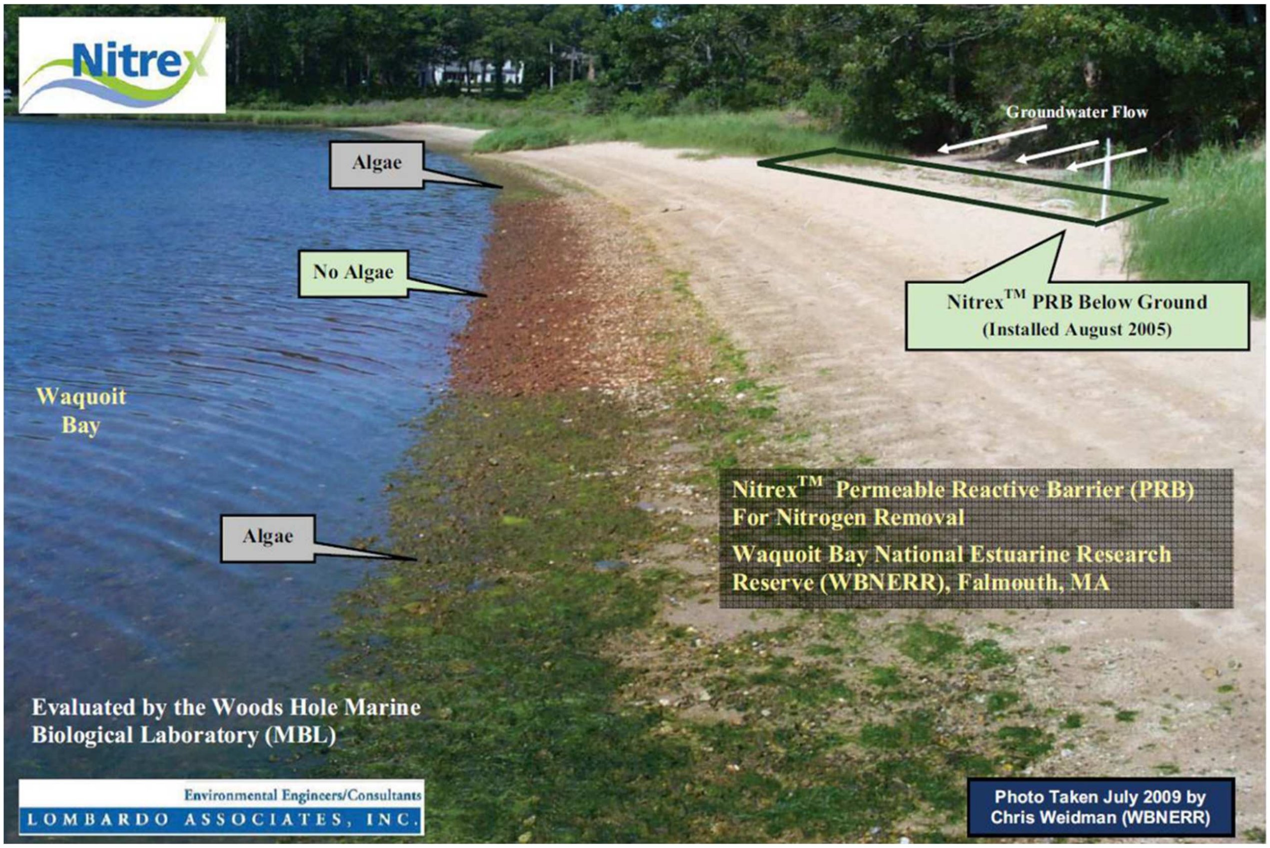

Figure 3. Installation of PRB on Cape Cod. Installation of one of the two pilot sized PRBs in the Cape Cod study. Easily seen is the algal difference between where the PRB is installed and where it is not.

Nitrate levels were reduced from approximately 450 µM (27.9 mg/L) to below 100 µM (6.2 mg/L) in the area of the PRB installation. There was a deeper nitrate contamination plume which passed under the PRB. This could occur anywhere and is why it is necessary to ensure hydraulic conductivity is higher in the PRB than the surrounding area.

Study Area

The Forge River lies between the hamlets of Mastic and Moriches, in southern Suffolk County, NY. It is a naturally shallow, partially mixed estuary of Moriches Bay (Swanson et al. 2008), cutting through the glacial outwash plain deposited by the Wisconsin glacial event (Swanson et al. 2009a). The Forge River has a tidal range of approximately one meter at Moriches Inlet. Forge River flow has been altered by man in the recent past. Being naturally shallow, it has been dredged several times to remove duck waste sludge in order to control pollution, as well as for navigational purposes (Swanson et al. 2008). The Great South Bay into which the Forge River flows has also been altered by natural events in recent history, opening and closing inlets and breaches (Morang 1999).

As a major tributary of Moriches, the Forge River has both commercial and recreational value for the surrounding population, although impairments of water quality have been an issue for over a century. In the 1950s, the Woods Hole Oceanographic Institute (WHOI) considered its tributaries to be extremely polluted (Redfield 1952), referring to the Forge River as “objectionable” and “highly contaminated” (Swanson et al. 2009a). In the first half of the twentieth century, contamination of the Forge River was attributed to duck ranches along the banks of the river which had started as far back at 1880s. Duck ranches were required to start regulating discharge in the 1950s. All ranches have now closed.

A rapid increase in population is causing part of the pollution problem. Between 1960 and 2005 there was almost a sevenfold population increase in the hamlets surrounding the Forge

River, from a population of 8,952 to a population of 59,000 (Swanson et al. 2008). Now, most nitrate contamination in the Forge River area is attributed to fertilizer runoff as well as septic-tank and cesspool discharge into the groundwater (Flipse et al. 1984). In general on Long Island, rural agriculture has declined, but it has been replaced with turf-grass fertilization on residential lawns (Varekamp et al. 2014). In addition, the surrounding area uses septic tanks for sewage remediation rather than sewage treatment plants. On Long Island, over half of residential areas are connected to septic systems. There are over 400,000 individual, on-site septic systems in Suffolk County alone, representing up to 75% of the population's domestic wastewater treatment (Twarowski et al. 2012). Many of these are likely to be outdated because homeowners tend to only pump or upgrade their septic system when they are faced with complications such as a backup (Twarowski et al. 2012). Groundwater nitrate that originates from septic systems can spread quickly and has been found in concentrations above the drinking water limit of 10 mg/L more than 100 m from small septic systems (Robertson et al. 1991).

Continuing pollution is therefore still a cause of algal blooms and in turn fish kills and foul odors (Swanson et al. 2008). Since 2005, the Forge River has suffered seasonally chronic hypoxia due to excessive nitrogen input from a number of natural and anthropogenic sources, including submarine groundwater discharge that contains effluent from unsewered high-density residential housing. An alarming fish-kill occurred in the summer of 2006, which increased the concern about a general decline in its state of health and, consequently, the Forge River was added to the 2006 New York State 303(d) List of Impaired Water Bodies. The Town of Brookhaven is looking for a solution to the nitrate loading into the surface waters of the area through submarine groundwater discharge. Permeable reactive barriers many be part of that solution.

Previous Work

The Forge River surface waters have already been studied and are a concern for the Town of Brookhaven, the United States Geological Survey and School of Marine and Atmospheric Sciences of Stony Brook University among others. In 2008, a paper was published outlining the varying problems of the Forge River's pollution (Swanson et al. 2008). Subsequently, papers were produced specifically dealing with the Forge River ecology (Swanson et al., 2009b), nutrients (Swanson et al., 2009c), and sediments (Bronwawell et al., 2009). The nutrient paper looked into the nitrogen contamination and the elevated levels of chlorophyll and phytoplankton biomass. That study showed that there was chlorophyll-α levels ranging from 10 to greater than 600 µg/L. The chlorophyll-α levels peaked in the summer and fall months. The study identified multiple sources of nitrogen entering the Forge River system and concluded that minimizing the impacts that the cesspools and septic tanks have on nitrogen entering groundwater would greatly lessen the nitrogen load to the River and may reduce the hypoxia of the Forge.

The Forge River sediments were found to be fine grained, and have an "incredibly high" total organic carbon (TOC) and nitrogen content (Bronawell et al. 2009). This is likely controlled by the "hyper-eutrophication" of the Forge River (Brownawell et al. 2009). Organic nitrogen levels were also found to be very high in the Forge River sediments.

In a study by Durand and Paulsen (2014) and Durand et al. (2011), a transect of the Wills Creek showed a high influx of nitrate at approximately 1.5 m below the surface of the creek represented by values of 21.7 mg/L and 22.4 mg/L (at the left of Figure 4) and, perhaps a deeper source at 3.5 m represented by a value of 15.6 mg/L (at the left of Figure 4) This influx however dissipates before 20 m away from the shore. This could indicate that the nitrate is being used by different species of algae and other marine species for growth.

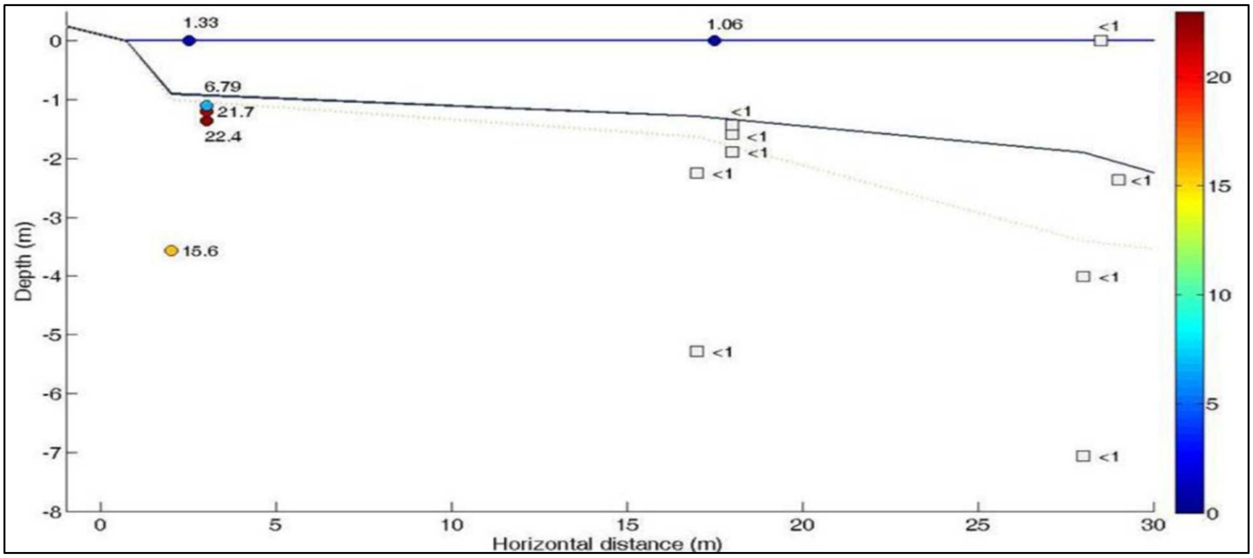

Figure 4. Nitrate Transect from Wills Creek. Data show an influx of nitrate in mg/L with a high at approximately 1.5 m and 3.5 m below the surface of the creek's north (left) shore. Horizontal axis is horizontal distance in meters from north shore of Wills Creek. From Durand et al. 2008, and Durand and Paulsen 2014.

Durand et al. (2008) and Durand and Paulsen (2014) also investigated electrical conductivity of the sediments at the Wills Creek shoreline. Electrical conductivity is a proxy of groundwater salinity (Stieglitz et al. 2008; Taniguchi et al. 2007; Weinstein et al. 2007) with more saline groundwater indicated by higher conductivity. Electrical conductivity measurements made by Durand et al. (2008) and Durand and Paulsen (2014) showed that the high nitrate concentrations were associated with a plume of fresh groundwater coming under the shoreline at a depth of about 1.5 m (blue area in Figure 5). It may be that submarine groundwater discharge creates substantial direct nutrient contamination compared to other sources, such as surface runoff or atmospheric deposition.

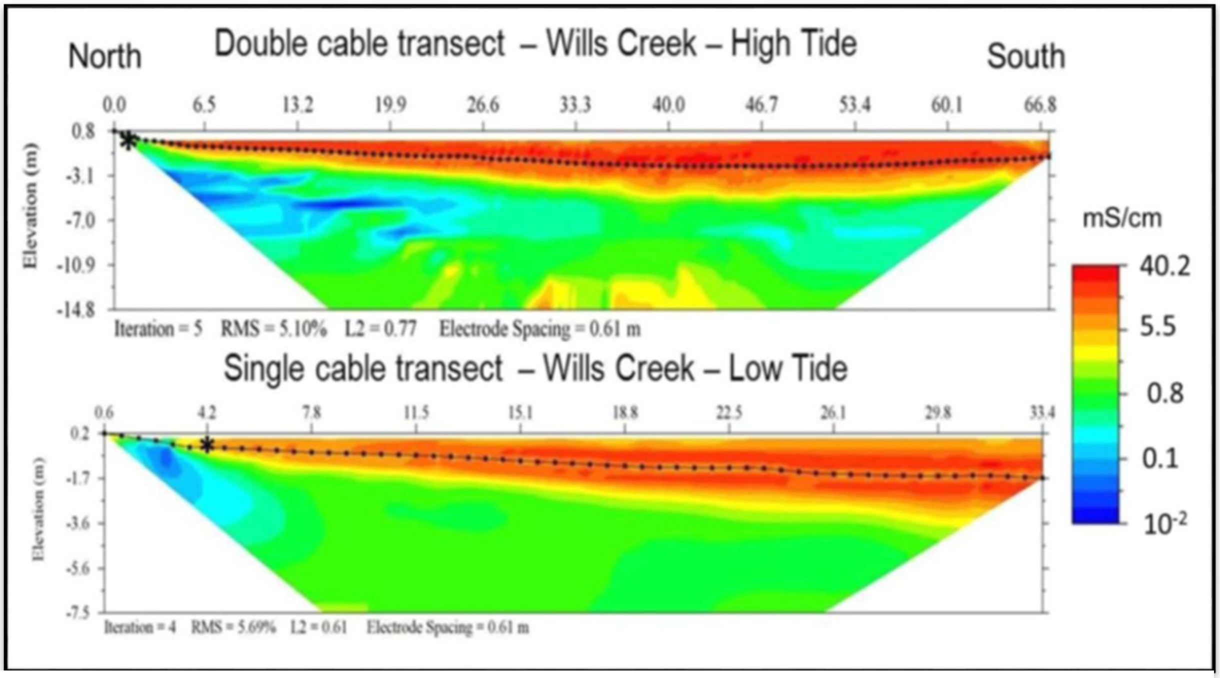

Figure 5. Conductivity Transect from Wills Creek. Data show an influx of fresh water approximately 1.5 m below the surface of the creek (seen in blue).Taken from Durand et al., 2008, and Durand and Paulsen, 2014.

Methods

Monitoring wells were installed at the study site. Several groundwater sampling events looking at nitrate-nitrite, ammonium, total nitrogen, and total organic carbon were performed in conjunction with water quality parameter collection for dissolved oxygen and conductivity. A rising well head test was done to determine hydraulic conductivity in the area.

Well Installation

A notice was sent to the residents of 62 Riviera Drive down to 72 Riviera Drive that monitoring well installation would occur and that noise level may be elevated in the area (Appendix 1). Six single-level monitoring wells were originally put in on the North side of Riviera Drive. The original six wells, MW-1 through MW-6, were installed via the direct push method by Paul Recchia of Tristate Geotechnical Drilling. The direct push method was chosen due to the depth of the water table being very shallow. It is considered the best method because of the sandy composition of the surrounding soil (Aller, 1991).

After checking the positions of utilities, the first 1.2 m were hand cleared and the next 1.8 m were direct pushed with the PVC wells inside the direct-push rods to create seven wells of 3 m in depth. When the rods were pulled from the hole, the hole was then backfilled with sand and a bentonite seal was placed from 0.3 to 0.6 m below grade (Figure 6). The manhole cover was cemented in place, flush with the surrounding grade (Lapham et al. 1997). Ten well-volumes were removed by pumping in order to clear any sediment that may have accumulated in the well casing (Striggow et al. 2013).

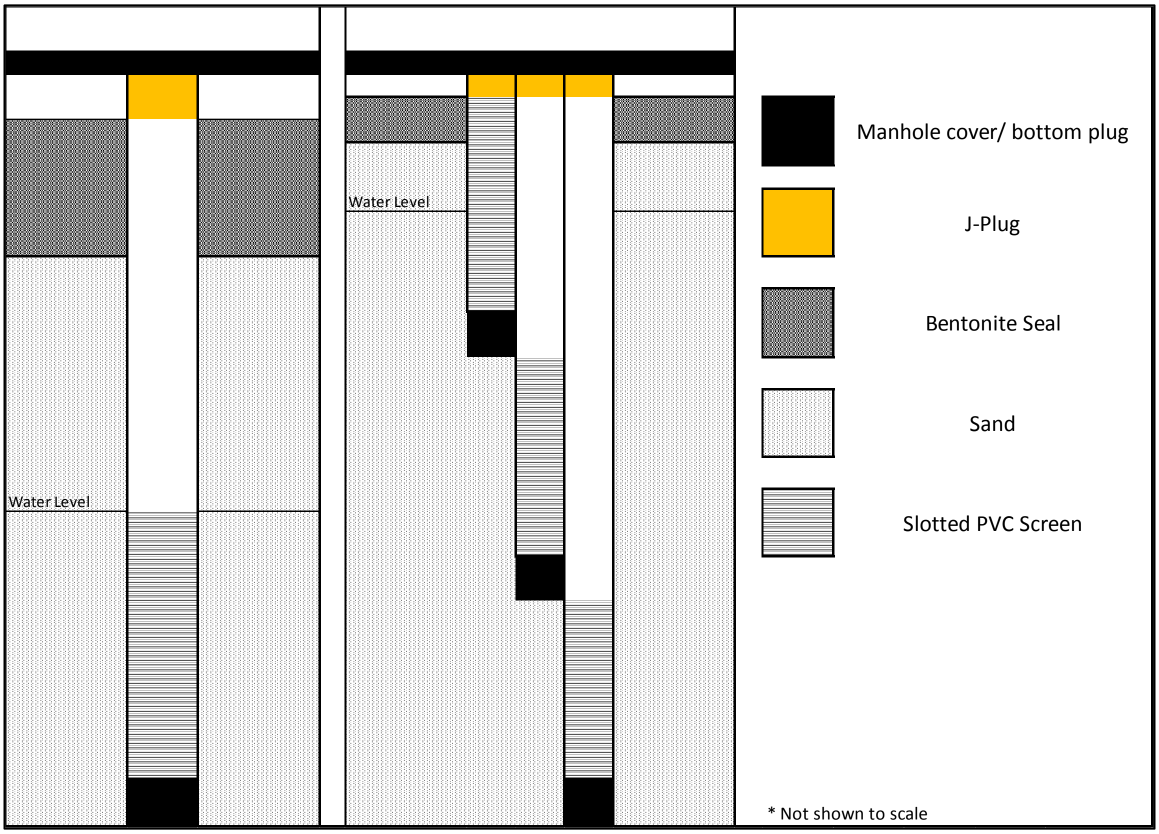

Figure 6. Construction of Monitoring Wells. Approximate single level well construction details on the left, multilevel well construction on the right.

Before installing the multilevel wells, one round of sampling was performed in order to place the multilevel wells appropriately. Two multilevel wells were subsequently put in with the hollow-auger drilling method. This type of drilling consists of a hollow steel stem with a continuous steel flight and carbide teeth in the auger bit (Striggow et al. 2013).

In accordance with Environmental Protection Agency drilling guidelines (Aller 1991), the first 1.2 m were hand cleared. After the first 1.2 m the well was over-drilled using a hollow stem auger approach to approximately 5 m. The borehole is over-drilled in order to more easily place the wells at depth. The three separate PVC wells of 1.5 m, 3.0 m, and 4.6 m were bound together and inserted into the augers, the bottom wood plug was knocked out, and the augers pulled out. The auger hole was then backfilled with sand and a bentonite seal was placed from approximately 0.5 m below grade (Figure 6). The manhole cover was cemented in place flush with the surrounding grade (Lapham et al. 1997). All levels of the multilevel wells were cleared of sediment by over pumping (Striggow et al. 2013).

Groundwater Sampling

The groundwater monitoring wells were sampled using a peristaltic pump and Teflon tubing. Teflon tubing was used in order avoid contaminating the samples (Schuller et al., 1981). It has been shown that pump materials can react with specific contaminants, which may affect water quality parameters (Powell and Puls, 1993). Tubing was placed in the monitoring well to the bottom of the casing.

Prior to sampling, each well was purged to remove stagnant water in accordance with United States Geological Survey guidelines (Lapham et al. 1997). The stagnant water was removed in order to obtain a sample representative of the area's groundwater. The water was then filtered through a GFF 45 micron filter and placed in 15 mL falcon tubes for nitrate-nitrite and ammonium analysis. For total nitrogen analysis the sample was also filtered through a GFF 45 micron filter and then placed in a 30 mL glass vial that was pre-preserved with hydrochloric acid in order to lower the pH of the sample to less than 3. The samples were then frozen to protect integrity until they were able to be analyzed.

Water from each well was also analyzed for water quality variables. A YSI 85 had been calibrated according to manufacturer's specifications and used to analyze a water sample from each well for temperature, conductivity and dissolved oxygen. Tubing was placed to the bottom of the well and after purging the stagnant water as described above, the pump was turned down and water was pumped into a sample cup. The cup was rinsed with sample water and then filled again, with the tubing releasing the water to the side of the cup to avoid splashing of the water as well as oxygen being added to the sample. The YSI was then submerged in the sample cup and allowed to sit out of the sun until readings equalized.

Sample Analysis

Samples were kept frozen between collection date and the day analysis was run. Nitrate-nitrite and ammonia were both analyzed using an automated flow injection colorimeter (Shinn, 1941). Because the fraction of nitrite is typically low, the combined nitrate-nitrite values are dominated by nitrate. This analysis is done using a standard sulphanilamide technique for nitrate-nitrite (Henriksen and Selmer-Olsen 1970), and a phenol technique for ammonia (Weatherburn 1967). The instrument used was a Lachat 6000.

Total nitrogen samples were first boiled to achieve high temperature oxidation (American Public Health Association 1915). The samples were then run through a chemoluminescent detector. The instrument the total nitrogen samples were run through is a Shimadzu TOC-5000.

Variable Head Well Test

In order to investigate the hydraulic conductivity of the underlying groundwater, a variable head test was performed. This area was assumed to have a high hydraulic conductivity, so a rising head well test was performed (Sara 2010). One monitoring well (MW-3 see Figure 7) was purged until significant draw down was achieved. Depth to water was then measured from the top of the well casing to water level, starting level and every fifteen seconds until water level had stabilized. The rising well head test was repeated six times. The average overall velocity was taken and used to determine the hydraulic conductivity of the surrounding aquifer (Bagarello et al. 2012).

The hydraulic conductivity, K, was calculated as (after Bouwer and Rice 1976):

$$\boldsymbol{K = \frac{\ln(\frac{R_e}{r_w})r_c^2}{2L}\,\frac{dln(y)}{dt}} \qquad [1]$$

where t is time in seconds, L is the screened length of the well, R_e is the radial distance over which the head lose is dissipated by the flow, r_w is the radial distance between the well center and the undisturbed aquifer outside of any well packing, r_c is the radius of the well itself, y_o is the initial height of the water level in the well above the undisturbed water table at the start of the test, and y_t is the measured water level in the well above the undisturbed water table at time t during the test. The term $\boldsymbol{\frac{dln(y)}{dt}}$ is found by graphing the change in the natural logarithm of the water height versus time and finding the slope of the line. The term $\ln(R_e/r_w)$ was calculated as (Bouwer and Rice 1976):

$$\ln\left(\frac{R_e}{r_w}\right) = \left[\frac{1.1}{\ln(H/r_w)} + \frac{A + B\ln(D-(H/r_w))}{(L/r_w)}\right]^{-1} \qquad [2]$$

where *D* is the distance from the undisturbed water table to the base of the permeable aquifer, H is the distance between the undisturbed water table and the bottom of the screened section of the well, and *A* and *B* are empirical coefficients that had been determined in general as a function of L/r_w previously by analog modeling (see Figure 3 in Bouwer and Rice 1976).

Results

Six monitoring wells (MW-1 to MW-6; Table 1), each five centimeters in diameter, were installed along the north side of Riviera Drive on 28 April, 2014 (Figure 7, Appendix 2). On June 10, 2014, two multilevel monitoring wells (ML-1and ML-2; Table 1), each 2.5 cm in diameter, and one, 5 cm monitoring well (MW-7; Table 1) were installed (Figure 5, Appendix 2). Monitoring wells were sampled initially on May 20, 2014. After the multilevel monitoring wells were installed, all were sampled on June 20, July 3, 9, 11, 14 and 18, 2014.

Well ID	Latitude	Longitude	Distance between wells
MW-7	40°47'43.79"N	72°49'59.04"W	----------------------------
ML-2	40°47'43.85"N	72°49'58.43"W	13.0 m (MW-7 to ML-2)
MW-1	40°47'43.79"N	72°49'58.10"W	7.7 m (ML-2 to MW-1)
MW-2	40°47'43.80"N	72°49'57.59"W	11.8 m (MW-1 toMW-2)
MW-3	40°47'43.80"N	72°49'57.10"W	13.4 m (MW-2 toMW-3)
ML-1	40°47'43.83"N	72°49'56.68"W	9.4 m (MW-3 to ML-1)
MW-4	40°47'43.79"N	72°49'56.50"W	8.3 m (ML-1 to MW-4)
MW-5	40°47'43.76"N	72°49'55.97"W	8.8 m (MW-4 toMW-5)
MW-6	40°47'43.74"N	72°49'55.46"W	12.0 m (MW-5 toMW-6)

Table 1. Monitoring Well Locations. The latitude and longitude locations of all wells installed at the Wills Creek site from west to east.

Figure 7. Locations of Installed Wells. Monitoring wells (MWs) and multilevel wells (MLs) along Riviera Drive on Wills Creek. Yellow circles represent locations of single level wells, red circles represent multilevel wells. Wells are located between 7.7 m and 13 m. (Image from Google Earth 2014)

Dissolved Oxygen

Variations in dissolved oxygen concentrations in each well over time tended to be smaller than differences among wells so that a spatial trend was observed (Figure 8). Dissolved oxygen concentrations were low throughout, with slightly higher values being found in the center of the transect (MW-2 and MW-3), near the east end (MW-5 and MW-6) and in west (MW-7). Higher values were also found in the multilevel wells, ML-1(15)[1] and ML-2(15), at a depth of 4.6 m (Figure 8).

Dissolved oxygen concentrations ranged from 0.52 mg/L at a depth of 1.5 m in ML-1 on May 20 to 4.08 mg/L in MW-7 on July 9, 2014 (Table 2). Over 80% of the samples were hypoxic (<3 mg/L). Only MW-5 and MW-3 reached levels of dissolved oxygen above 3 mg/L with any consistency, but even these never rose above 4 mg/L. Monitoring wells MW-5, MW-3, MW-2, ML-2 (at a depth of 4.5 meters), and MW-7 had levels of dissolved oxygen above 2.0 mg/L during all sampling events. The average dissolved oxygen concentrations over the sampling period showed low dissolved oxygen (less than 2.0 mg/L) in MW-1, MW-4, ML-1(5), ML-1(10), ML-2(15), and ML-2(10). The sampling events of June 20 and July 14 found average levels under 2.0 mg/L due, in part, to low concentrations of dissolved oxygen at and near MW-1 and the shallowest depth in ML-1.

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[1] The multilevel wells were installed at three depths 1.5m, 3.0m, and 4.6m. These were identified as ML-1(5), ML-1(10), and ML-1(15) were the 5, 10, and 15 refer to the depths in feet.

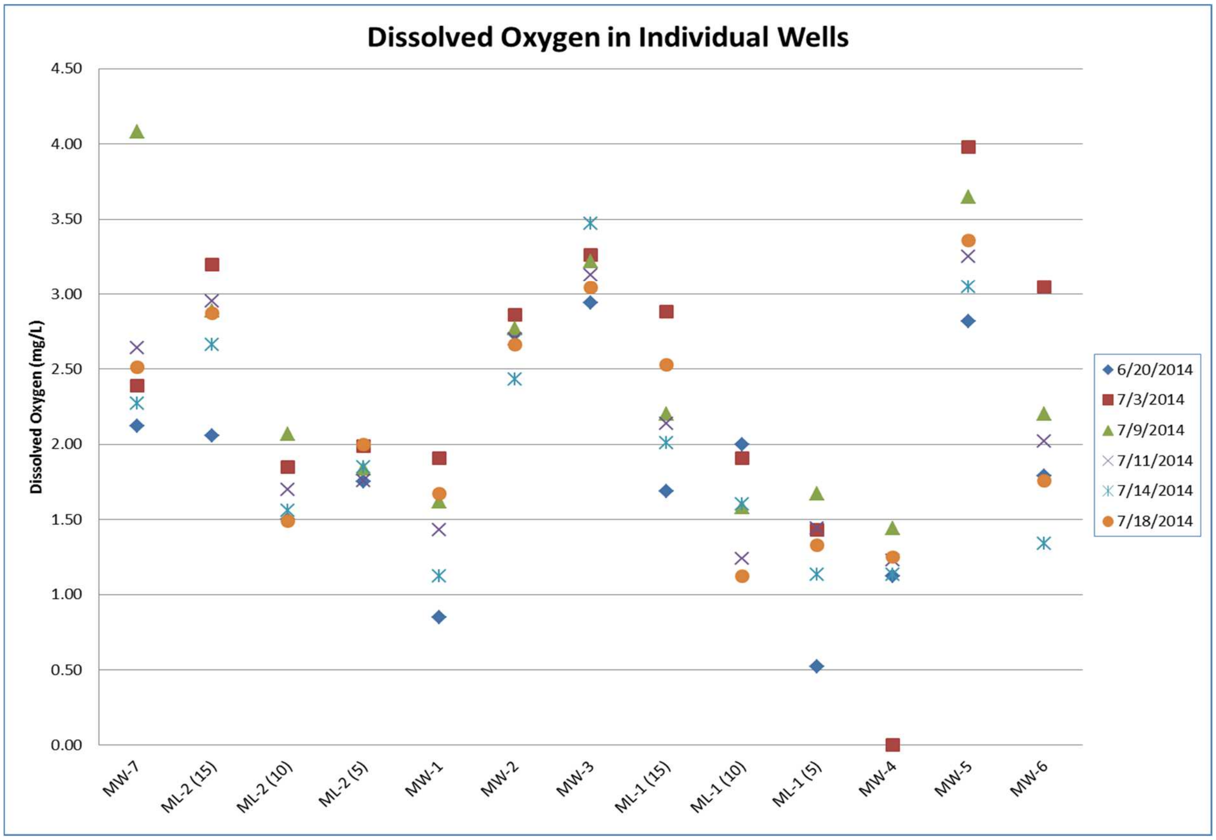


Figure 8. Dissolved Oxygen in Individual Wells. Wells are shown in the order their location west to east across the study area. Dissolved oxygen is shown in mg/L.

	6/20/2014	7/3/2014	7/9/2014	7/11/2014	7/14/2014	7/18/2014	**Well Average over Study Period**
MW-6	1.79	3.05	2.20	2.02	1.34	1.76	**2.03**
MW-5	2.82	3.98	3.65	3.25	3.05	3.36	**3.35**
MW-4	1.12	--	1.44	1.23	1.13	1.25	**1.23**
ML-1 (5)	0.52	1.43	1.67	1.44	1.13	1.33	**1.25**
ML-1 (10)	2.00	1.91	1.58	1.24	1.60	1.12	**1.58**
ML-1 (15)	1.69	2.88	2.20	2.14	2.01	2.53	**2.24**
MW-3	2.94	3.26	3.22	3.13	3.47	3.04	**3.18**
MW-2	2.73	2.86	2.77	2.70	2.43	2.66	**2.69**
MW-1	0.85	1.91	1.62	1.43	1.12	1.67	**1.43**
ML-2 (5)	1.75	1.99	1.84	1.76	1.85	2.00	**1.87**
ML-2 (10)	1.50	1.85	2.07	1.70	1.56	1.49	**1.70**
ML-2 (15)	2.06	3.20	2.89	2.95	2.66	2.87	**2.77**
MW-7	2.12	2.39	4.08	2.64	2.27	2.51	**2.67**
Average for each Sampling Event	**1.84**	**2.56**	**2.40**	**2.13**	**1.97**	**2.12**	

Table 2. Dissolved Oxygen Data in mg/L. Wells are shown in order that they are located geographically from west to east across the study area. Dashed lines indicate a well was not able to be sampled.

Low levels of dissolved oxygen (< 2 mg/L) were seen in two areas (Figure 8). The westernmost area was detected in wells ML-2 (at a depth of 1.5 m and 3.0 m) and MW-1. Another area of low dissolved oxygen was found further to the east in wells ML-1 and MW-4 (Figure 8). These indicated two plumes of poorly oxygenated groundwater. The westernmost plume had a width between 7.7 m, the distance between ML-2 and MW-1, and 32.5 m, the distance between MW-7 and MW-2. The easternmost plume had a width between 8.3 m, the distance between ML-1 and MW-4, and 26.5 m, the distance between MW3 and MW-5. In the multilevel wells, the shallow wells, at a depth of 1.5 m, had lower dissolved oxygen concentrations than the deep wells (depth of 4.5 m), suggesting that plumes of low dissolved oxygen concentrations were confined to within three meters depth. The lowest average levels of dissolved oxygen were seen on June 20 and the highest on July 3, 2014, but there did not seem to be a trend with time for dissolved oxygen. Dissolved oxygen concentrations were found to be most variable in MW-6, where the concentrations ranged from 1.34 mg/L on July 14, to 3.05 mg/L on July 3. This may be the western edge of a plume of low-oxygen groundwater further to the east, beyond the study area.

Conductivity

Groundwater conductivity is generally a proxy for salinity with more saline groundwater having a higher conductivity. Seawater has a conductivity of about 56,000 micro-Siemens/cm (µS/cm), tap water typically ranges from 500 µS/cm to 800 µS/cm. Conductivity ranged from 111.1 µS/cm in MW-5 on July 3 to 1617.0 µS/cm in ML-1 (1.5m) on June 20, 2014 (Table 3). There does not seem to be a trend with time or space. In the multilevel wells there is higher conductivity in the shallow wells, depths of 1.5 m than elsewhere. ML-1(5) had an average of 1025.4 µS/cm and ML-2(5) had an average of 745.8 µS/cm over the span of the study period. All other wells had study period averages under 550 µS/cm. This difference in conductivity could be due to ML-1(5) and ML-2(5) being made entirely of screen with sediment contamination. The higher conductivity in the shallower wells may also be due to salt water contamination along the banks of the creek. Higher and lower conductivity may also be linked to both precipitation levels before a sampling event as well as tidal activity during the event. More research would be needed to show a link to either.

	6/20/2014	7/3/2014	7/9/2014	7/11/2014	7/14/2014	7/18/2014	**Well Average over Study Period**
MW-6	271.0	365.2	344.4	221.5	323.2	326.2	**308.6**
MW-5	119.0	111.1	148.5	162.7	151.8	160.9	**142.3**
MW-4	371.0	--	363.9	234.7	332.5	360.4	**332.5**
ML-1 (5)	1617.0	823.0	796.0	934.0	612.1	1370.0	**1025.4**
ML-1 (10)	340.0	312.1	361.6	290.5	281.8	332.2	**319.7**
ML-1 (15)	246.0	262.7	257.8	283.7	283.3	274.7	**268.0**
MW-3	286.0	274.1	267.1	268.0	269.7	271.3	**272.7**
MW-2	244.0	266.7	258.1	263.7	253.5	282.7	**261.5**
MW-1	451.0	554.0	541.0	540.0	542.1	539.1	**527.9**
ML-2 (5)	991.0	661.0	654.0	673.0	741.2	754.3	**745.8**
ML-2 (10)	284.0	267.0	325.3	246.1	242.3	243.2	**268.0**
ML-2 (15)	245.0	242.4	245.2	249.3	240.8	265.1	**248.0**
MW-7	196.0	252.4	207.6	279.0	237.2	216.7	**231.5**
Average for Each Sampling Event	**435.5**	**366.0**	**367.0**	**357.4**	**347.0**	**415.1**	

Table 3. Conductivity Data in µS/cm. Wells are shown in order of their location, west to east across the study area. Dashed lines indicate a well was not able to be sampled.

Nitrate-nitrite

Nitrate-nitrite levels ranged from 0.46 mg/L in ML-1(5) on July 18 to 22.95 mg/L in MW-1 on July 18 (Table 4). Concentrations of nitrate-nitrite consistently exceeded the drinking-water limit of 10 mg/L in well MW-1 and at the deepest level (4.6 m) of ML-1 (Figure 8); low concentrations of dissolved oxygen (near or below 2 mg/L) were also seen in ML-1. Low concentrations of nitrate-nitrite were seen with relatively high concentrations of dissolved oxygen in wells MW-7, MW-3 and MW-5.

Over the study period MW-1 had the highest average concentration of nitrate-nitrite at 17.14 mg/L. Monitoring well ML-1(15) also showed average nitrate-nitrite levels over drinking water limits (10 mg/L) with 14.27 mg/L over the study period. On average, only ML-2(15) showed nitrate-nitrite levels close to drinking water limits at 9.12 mg/L. In the multilevel wells, nitrate -nitrite concentrations increased with depth. The lowest concentrations of nitrate-nitrite were found in the shallowest depths in the multilevel wells (1.5 m), whereas the highest concentrations of nitrate-nitrite in the multilevel wells are found at the deepest levels at depths of 4.5 m. We may be looking at a water-table nitrate-nitrite plume at MW-1less than 20 m across, but a deeper plume at 4.5 m depth more widely distributed across the area.

In MW-1, MW-2, MW-3, ML-1(15), MW-6 and MW-7 there was a slight increase in wells over time. The last sampling date (July 18) had the highest level of nitrate-nitrite found in each of these individual wells (Figure 9). There tended to be a slight increase in levels of nitrate-nitrite with decreasing concentrations of dissolved oxygen (Figure 10), but dissolved oxygen levels tended to be low in all samples. Some wells had consistently low levels during every sampling event. In addition, the variability was fairly high. Some wells with higher dissolved oxygen had lower nitrate-nitrite levels, such as in MW-5. Monitoring well MW-5 had an average of 2.84 mg/L of nitrate-nitrite and 3.35 mg/L of dissolved oxygen over the study period. However, the reverse was also true; MW-1 had an average of 17.14 mg/L nitrate-nitrite and 1.43 mg/L dissolved oxygen.

	6/20/2014	7/3/2014	7/9/2014	7/11/2014	7/14/2014	7/18/2014	**Well Average over Study Period**
MW-6	2.23	4.92	7.51	1.59	3.41	6.67	**4.39**
MW-5	3.59	2.68	2.43	2.85	2.98	2.53	**2.84**
MW-4	2.82	--	3.3	3.16	2.6	1.84	**2.74**
ML-1 (5)	--	1.11	--	0.98	1.05	0.46	**0.90**
ML-1 (10)	7.87	9.45	6.23	6.65	8.02	8.13	**7.73**
ML-1 (15)	14.64	14.08	--	13.2	12.59	16.82	**14.27**
MW-3	3.8	2.02	1.65	1.75	1.82	4.66	**2.62**
MW-2	3.91	4.88	4.26	4.74	6.38	6.78	**5.16**
MW-1	10.86	18.35	11.1	18.79	20.8	22.95	**17.14**
ML-2 (5)	--	3.53	4.74	3.26	3.12	4.99	**3.93**
ML-2 (10)	6.57	5.54	5.25	5.98	5.87	6.74	**5.99**
ML-2 (15)	8.63	13.21	8.64	8.72	7.62	7.88	**9.12**
MW-7	3.06	3.75	2.98	3.31	4.08	4.96	**3.69**
Average for each Sampling Event	**6.18**	**6.96**	**5.28**	**5.77**	**6.18**	**7.34**	

Table 4. Nitrate-nitrite Data. All data shown in mg/L. Wells are shown in order that they are located west to east across the study area. Double dashed lines indicate a well was not able to be sampled. Multilevel wells and MW-7 were not installed until the June 20, 2014 sampling event.

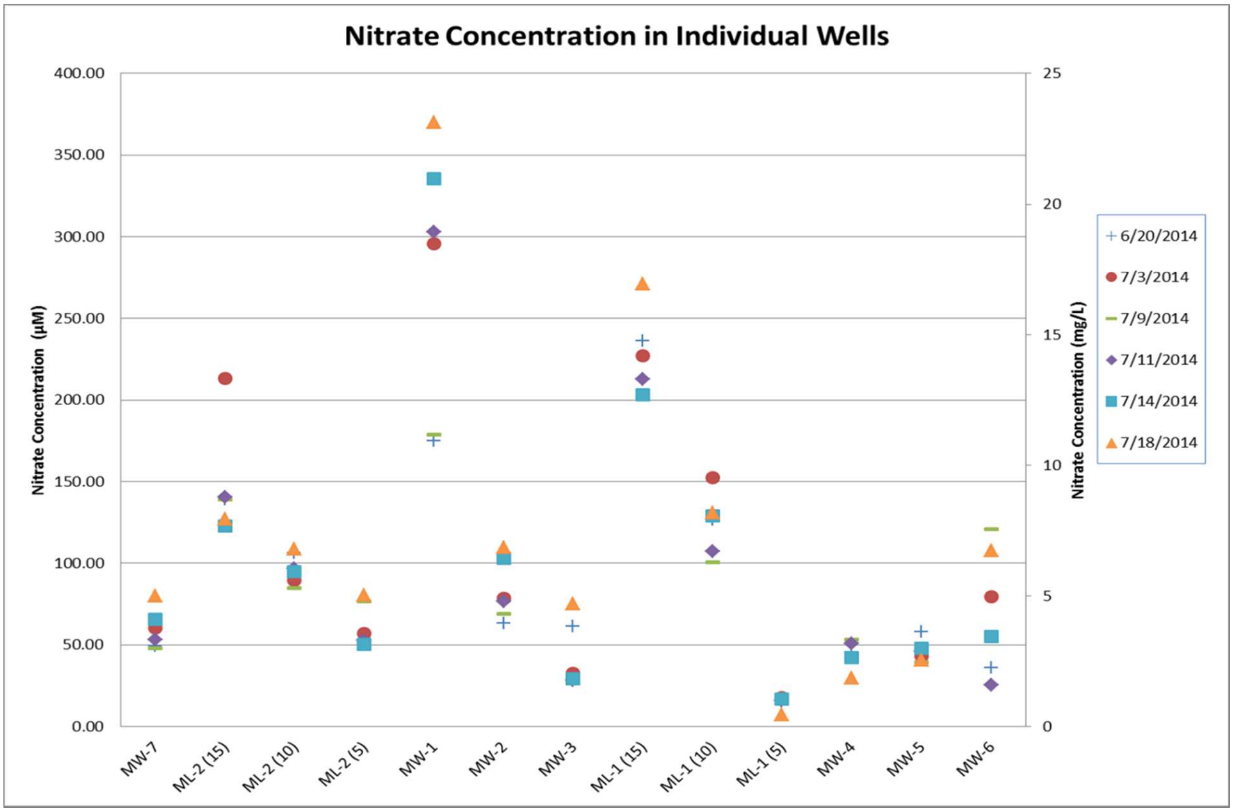


Figure 9. Nitrate-nitrite Concentrations in Individual Wells. Nitrate concentrations are shown in µM on the left vertical axis and mg/L on the right vertical axis. Wells are shown as they are spatially located from west to east across the study area.

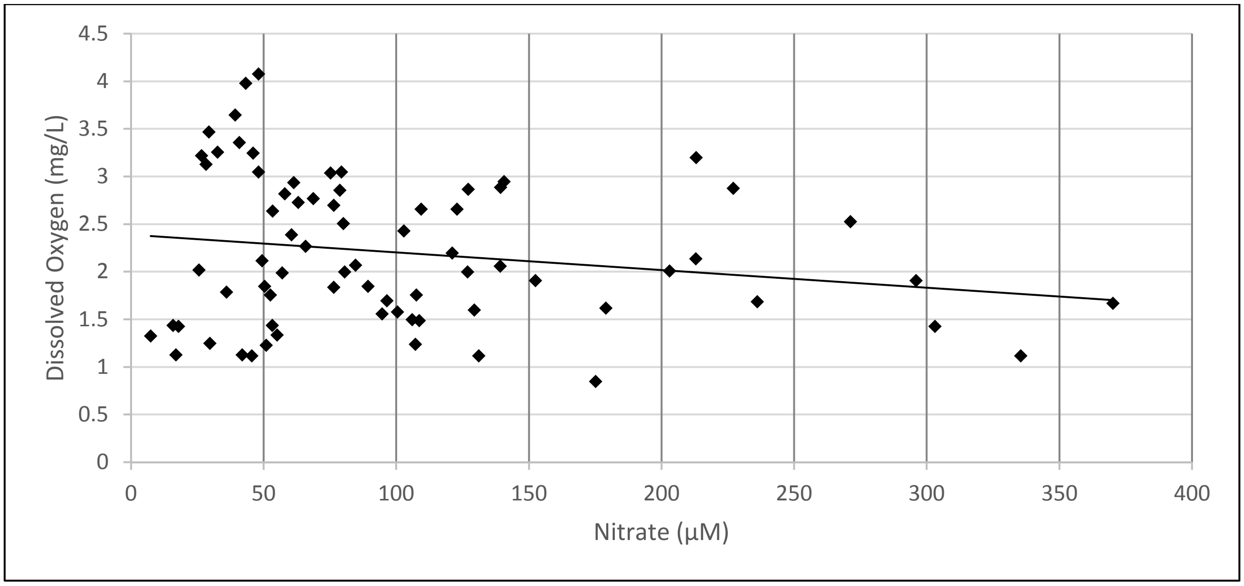


Figure 10. Dissolved oxygen concentrations versus Nitrate-nitrite concentrations showing a slightly negative linear correlation (DO= -0.002*(NO_3+NO_2^-)+2.4) with an R^2 of 0.036.

Ammonium

Ammonium concentrations tended to be low. Ammonium was not detectable (<0.01 mg/L) in 17% (12) of the samples and below 1 mg/L in 42% (31) more. In wells where there was very little ammonium or it was non-detectable, the levels stayed steady. Higher ammonium concentrations were also found in MW-1 and MW-4 with the highest values of 19.56 mg/L in MW-1 on July 9, 2014 (Table 5). There was an increase over time in MW-1 where the highest levels of ammonium where also detected over the entire study area (Figure 11). MW-1 showed an average of 11.80 mg/L and MW-4 showed an average of 8.14 mg/L over the course of the study. Apart from MW-1 and MW-4, only ML-1(10), ML-2(10) and MW-3 had an average ammonium level about 1.0 mg/L. All other wells averaged less than 0.5 mg/L of ammonium over the study period. The sampling event with the highest average ammonium was July 9 with an average of 4.13 mg/L of ammonium. Ammonium showed the highest variability in MW-4, from a low of <0.01 mg/L on July 18 to a high of 10.09 mg/L on June 20. The sampling event with the highest average ammonium was July 9 with an average of 4.13 mg/L of ammonium. Ammonium concentrations support the interpretation of two water-table plumes, one at MW-1 and another in the vicinity of MW-4.

	6/20/2014	7/3/2014	7/9/2014	7/11/2014	7/14/2014	7/18/2014	**Well Average over Study Period**
MW-6	0.01	0.03	<0.01	0.10	<0.01	<0.01	**0.05**
MW-5	<0.01	<0.01	0.19	0.20	0.07	0.04	**0.13**
MW-4	10.09	--	7.82	7.95	6.71	<0.01	**8.14**
ML-1 (5)	--	0.09	--	0.05	0.05	0.06	**0.06**
ML-1 (10)	2.65	3.76	2.12	2.30	2.68	2.24	**2.63**
ML-1 (15)	0.26	0.08	--	0.33	0.25	0.06	**0.20**
MW-3	0.13	0.04	<0.01	0.01	<0.01	8.67	**2.21**
MW-2	0.11	<0.01	0.01	<0.01	<0.01	<0.01	**0.06**
MW-1	4.21	10.93	19.56	10.73	12.36	13.03	**11.80**
ML-2 (5)	--	0.02	0.01	0.02	0.01	0.05	**0.02**
ML-2 (10)	3.37	3.09	3.29	3.32	2.85	2.47	**3.07**
ML-2 (15)	0.03	<0.01	<0.01	0.05	<0.01	<0.01	**0.04**
MW-7	0.03	0.45	0.01	<0.01	<0.01	<0.01	**0.16**
Average for each Sampling Event	**2.09**	**2.05**	**4.13**	**2.28**	**3.12**	**3.33**	

Table 5. Ammonium Data. All data shown in mg/L. Wells are shown in order that they are located west to east across the study area. ND indicates that levels in sample were not detected. Double dashed lines indicate a well was not able to be sampled.

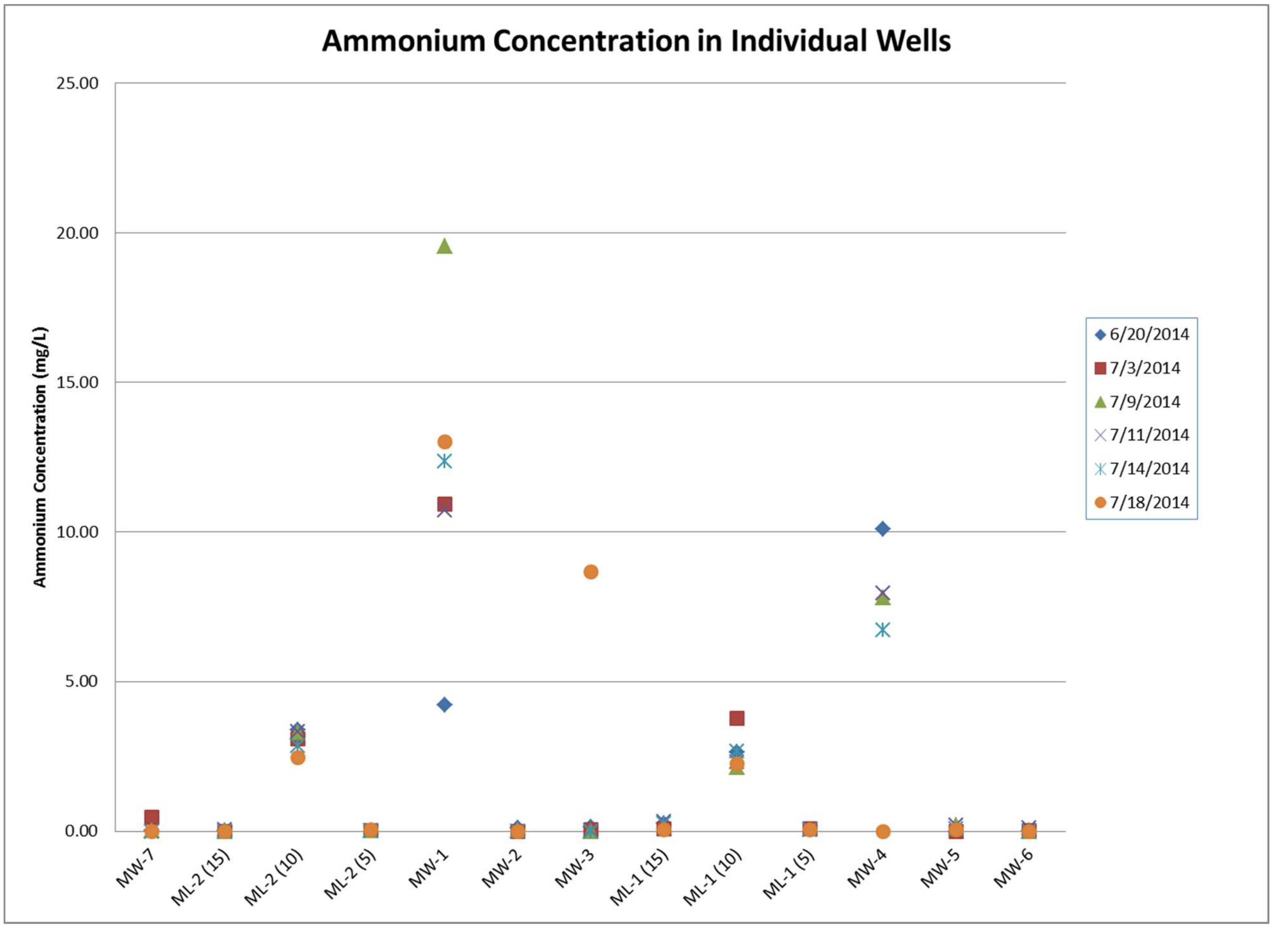


Figure 11. Ammonium Concentration in Individual Wells. Concentrations are in mg/L. Wells are ordered how they are located west to east across the study area.

Total Nitrogen

Total nitrogen ranged from 1.45 mg/L in MW-4 on July 18 to 39.00 mg/L in MW-1 on July 14 (Table 6). There is also no trend in the data in accordance with depth in the multilevel wells (Figure 12). On average over the study period, MW-1 had the highest total dissolved nitrogen with 23.95 mg/L. Concentrations exceeded 10 mg/L in MW-1 and MW-2, at the deepest level in ML-1(15) as well as once in mid-depth in ML-2(10). Monitoring well ML-1(15) was the only other well besides MW-1 with an average total dissolved nitrogen level over 10 mg/L with an average of 11.54 mg/L over the study period. The high level of total nitrogen in MW-1 could be attributed to the high level of ammonium also found in MW-1. As expected, these results seemed to indicate a contaminant plume intercepted by MW-1 at the water table, perhaps, another water-table plume near MW-2 and some evidence of a deeper contaminant plume more widely distributed across the study section.

	6/20/2014	7/3/2014	7/9/2014	7/11/2014	7/14/2014	7/18/2014	**Well Average over Study Period**
MW-6	2.18	2.70	5.31	2.16	4.48	6.96	**3.97**
MW-5	2.56	2.56	2.32	2.68	2.56	1.99	**2.45**
MW-4	11.44	--	10.16	8.19	6.26	1.45	**7.50**
ML-1(5)	--	1.71	2.94	2.35	--	2.31	**2.33**
ML-1(10)	3.38	5.04	6.67	6.93	4.19	8.90	**5.85**
ML-1(15)	10.47	6.35	10.74	15.01	12.35	14.32	**11.54**
MW-3	2.58	1.36	1.45	1.65	1.74	10.29	**3.18**
MW-2	3.63	2.01	3.73	4.60	3.68	2.22	**3.31**
MW-1	9.28	21.76	22.57	32.67	39.00	18.40	**23.95**
ML-2(5)	--	--	4.57	4.41	--	4.02	**4.33**
ML-2(10)	21.49	3.18	6.76	8.40	7.14	7.68	**9.11**
ML-2(15)	7.59	4.77	5.06	5.03	5.89	7.10	**5.91**
MW-7	2.51	2.17	1.73	2.51	3.20	3.49	**2.60**
Average for each Sampling Event	**7.01**	**4.87**	**6.46**	**7.43**	**8.23**	**6.86**	

Table 6. Total Nitrogen Data. All data is shown in mg/L. Wells are shown in order that they are located from west to east across the study area. Double dashed lines indicate a well was not able to be sampled.

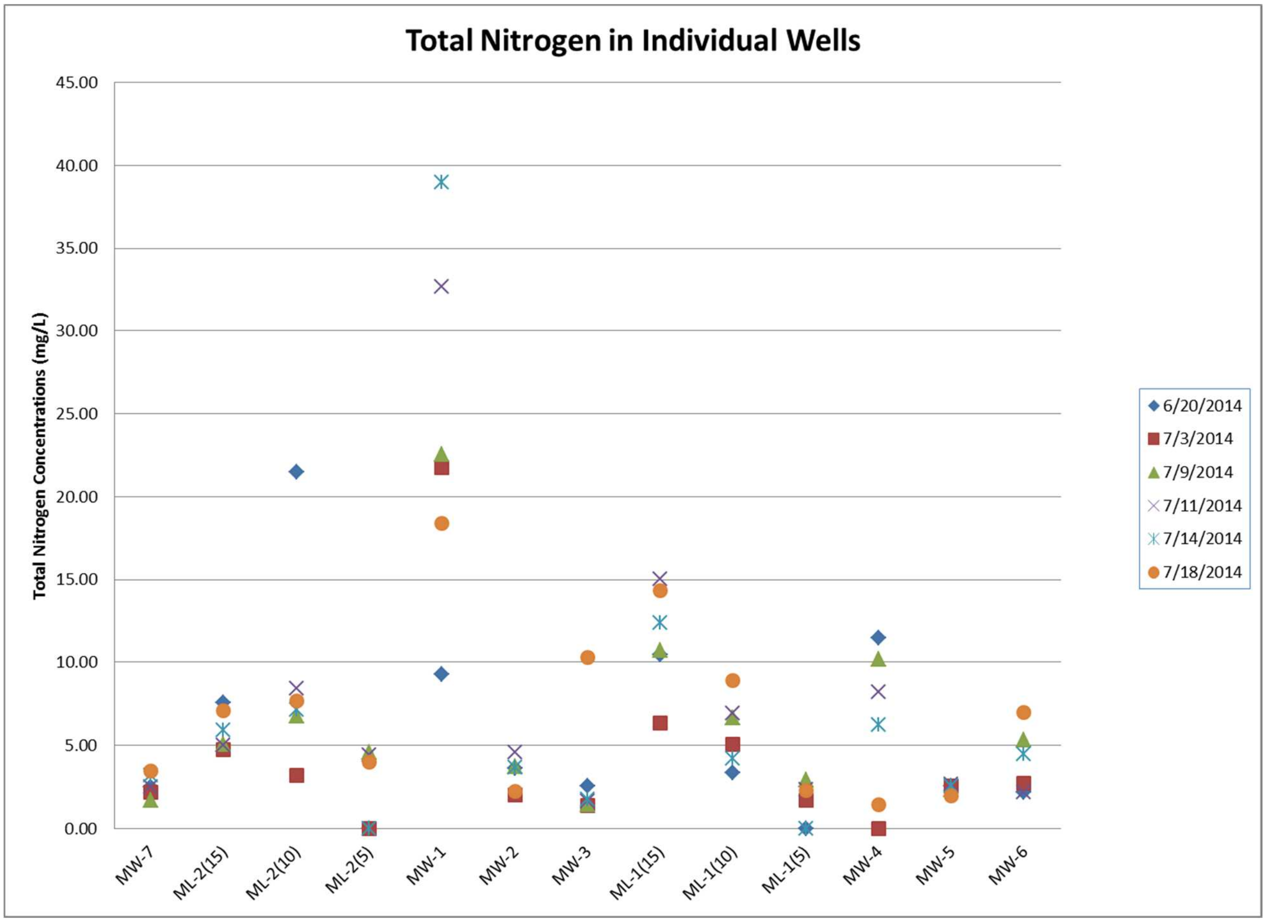


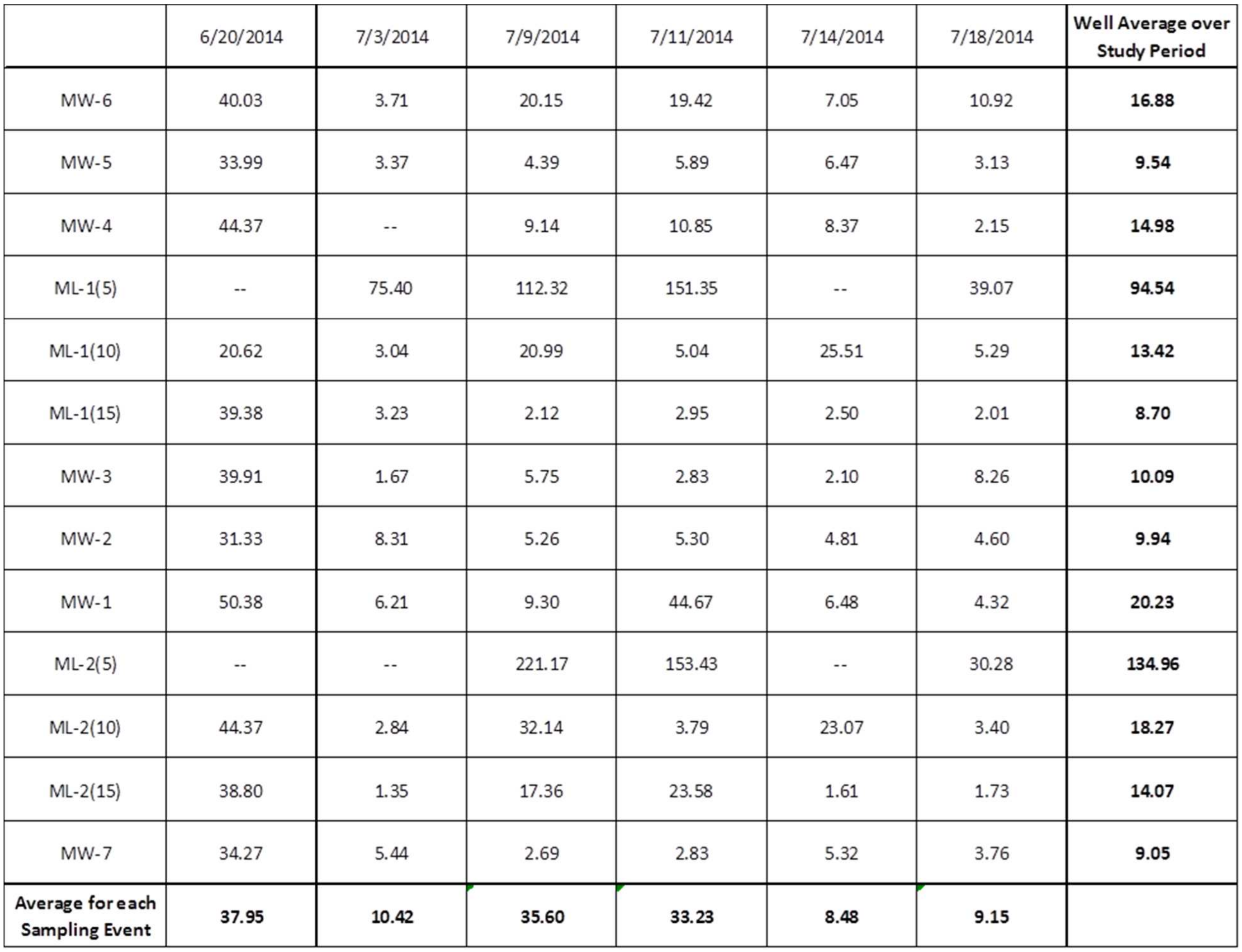
Figure 12. Total Nitrogen in Individual Wells. Total Nitrogen concentrations are in mg/L. The wells are placed on the figure as they are located west to east across the study area.

Total Organic Carbon

Total organic carbon ranged from a low of 1.35 mg/L in ML-2(15) on July 3 to a high of 221.17 mg/L in ML-2(5) on July 9 (Table 7). Total organic carbon concentrations remained below 50 mg/L (Figure 13) except at ML-1(15) and ML-2(5); higher total organic carbon was found at a depth of 1.5 m of the multilevel wells. Monitoring wells ML-1(5) shows an average of 94.54 mg/L and ML-2(5) shows an average of 134.96 mg/L over the study period. The rest of the monitoring wells show an average of less than 20 mg/L over the study period. The multilevel wells show a decreasing trend with depth (Figure 13).

The organic carbon content of the sampled groundwater was generally lower than 50 mg/L except at two locations (Table 7). Organic contents exceeded 100 mg/L at the water table in ML-1(5) and ML-2 (5). The highest value of 222.17 mg/L was found at ML-2(5) on July 9, 2014. As suggested by the distribution of other parameters, there seemed to be two groundwater plumes, one at ML-1(5) extending, to ML-4 and one at ML-2(5) reaching towards MW-1.

The individual sampling events were split where three events showed higher total organic carbon levels, and three showed lower levels. June 20, July 9, and July 11 all had an average of over 30mg/L of total organic carbon. The other three sampling events, July 3, July 14, and July 18 had levels around 10 mg/L of total organic carbon.



	6/20/2014	7/3/2014	7/9/2014	7/11/2014	7/14/2014	7/18/2014	**Well Average over Study Period**
MW-6	40.03	3.71	20.15	19.42	7.05	10.92	**16.88**
MW-5	33.99	3.37	4.39	5.89	6.47	3.13	**9.54**
MW-4	44.37	--	9.14	10.85	8.37	2.15	**14.98**
ML-1(5)	--	75.40	112.32	151.35	--	39.07	**94.54**
ML-1(10)	20.62	3.04	20.99	5.04	25.51	5.29	**13.42**
ML-1(15)	39.38	3.23	2.12	2.95	2.50	2.01	**8.70**
MW-3	39.91	1.67	5.75	2.83	2.10	8.26	**10.09**
MW-2	31.33	8.31	5.26	5.30	4.81	4.60	**9.94**
MW-1	50.38	6.21	9.30	44.67	6.48	4.32	**20.23**
ML-2(5)	--	--	221.17	153.43	--	30.28	**134.96**
ML-2(10)	44.37	2.84	32.14	3.79	23.07	3.40	**18.27**
ML-2(15)	38.80	1.35	17.36	23.58	1.61	1.73	**14.07**
MW-7	34.27	5.44	2.69	2.83	5.32	3.76	**9.05**
Average for each Sampling Event	**37.95**	**10.42**	**35.60**	**33.23**	**8.48**	**9.15**	

Table 7. Total Organic Carbon. All data shown in mg/L. Wells are shown in order that they are located west to east across the study area. Double dashed lines indicate a well was not able to be sampled. On June 20, 2014 ML-1 (5) and ML-2 (5) were unable to be sampled due to an overwhelming amount of sediment in the wells.

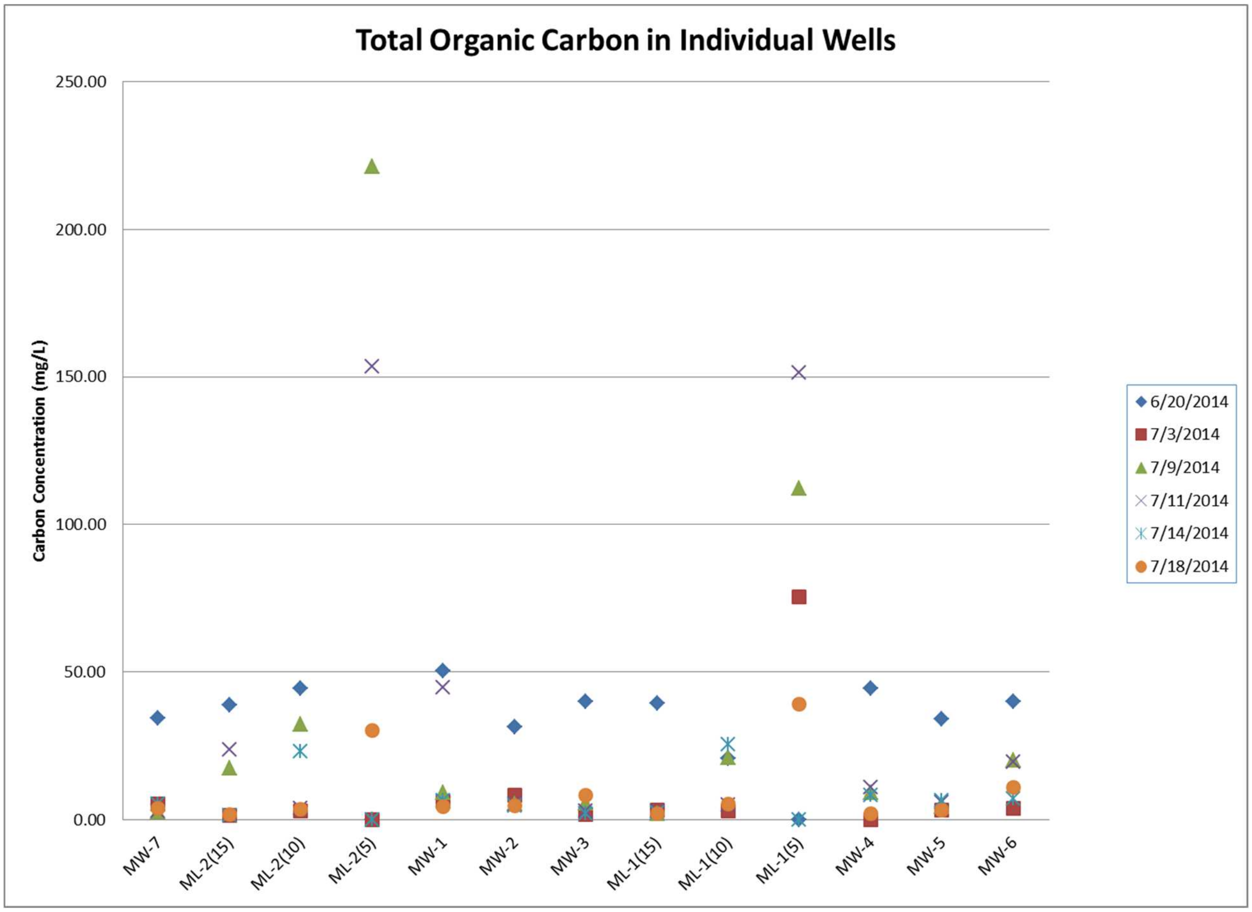


Figure 13. Total Organic Carbon in Individual Wells. Carbon concentrations are in mg/L. Wells are located on the figure as they are located west to east across the study area.

Rising-Head Well Test

During the rising-head well test (Figure 14) at MW-4, the well-water level rose at an average rate of 20 cm/min or approximately 0.0033 m/s (Table 8). The test was repeated six times with approximately the same results, and the mean value was used. Calculations are given in Appendix 3. The hydraulic conductivity was found to be 8.4 cm/d. This is so much smaller than other estimates of the hydraulic conductivity in the Upper Glacial aquifer, as to suggest that the test well was not representative of the ambient conditions. In the Upper Glacial Aquifer, hydraulic conductivity is expected to be above 8.2 m/day (McClymonds and Franke 1972).

Although the low hydraulic conductivity calculated here may be representative of a local aquitard, it seemed to me it may be an artifact of the well. I would suggest that an alternate method be used at another well, for further study.

	rep 1	**rep 2**	**rep 3**	**rep 4**	**rep 5**	**rep 6**
time	1423	1430	1438	1454	1459	1305
Start	1.54	1.58	1.5	1.52	1.63	1.67
15sec	1.4	1.42	1.33	1.37	1.47	1.52
30sec	1.19	1.23	1.17	1.2	1.29	1.35
45 sec	1.2	1.25	1.17	1.2	1.21	1.21
60 sec	--	--	--	--	1.25	1.2
original	1.21	1.22	1.2	1.22	1.24	1.22

Table 8. Rising Well Head Data. Data from rising well head test performed MW-4 on July 20, 2014. The test was repeated six times on the same day, on the same well.

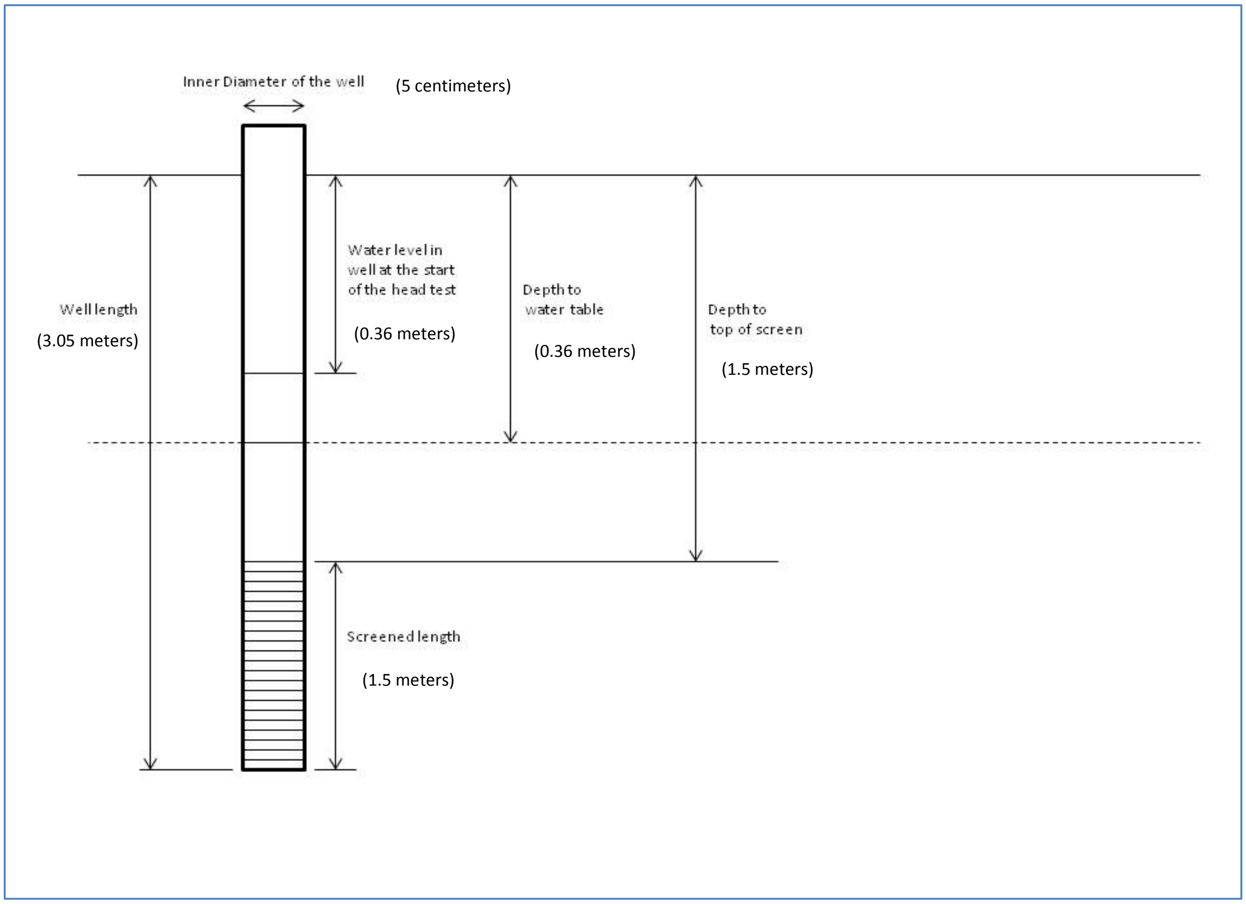


Figure 14. Variables measured for Rising-head Well Test.

Discussion

The purpose of this study was to determine the acceptability of Wills Creek for a permeable reactive barrier for removal of groundwater nitrate contamination. In order to be considered a candidate for a permeable reactive barrier, it is necessary for the site to have high nitrate concentrations and low dissolved oxygen concentrations.

Based on levels of dissolved oxygen, nitrate-nitrite concentration and ammonium concentrations, and organic carbon content, water table plumes were seen in two locations. A region in the vicinity of MW-1 was characterized by nitrate-nitrite concentrations averaging

17.14 mg/L, dissolved oxygen levels averaging 1.43 mg/L, concentrations of ammonium averaging 11.8 mg/L and organic carbon concentration of 20.23 mg/L. This plume would have a width between 7.7 m, the distance between ML-2 and MW-1, and 32.5 m, the distance between MW-7 and MW-2. The evidence for a water table plume in the vicinity of MW-2 was less compelling. The variability of ammonium concentrations in MW-2 suggested to me that the well was at the margins of a plume. Since it was not well-detected in MW-3 on MW-5, a plume here would be less than 26.5 m wide. The limited extent and spacing of these plumes, as well as their location in front of individual houses, likely mean that they were linked to the location of the individual septic systems of the houses.

Although the average concentrations of nitrate-nitrite in the deeper levels of both the multilevel wells were not as high as those found at the water table in MW-1, nitrate-nitrite levels were elevated in both ML-1(15) and ML-2(15). I interpret this as evidence of a more dilute, but deeper contaminant plume passing under the entire area due to nitrate-nitrite entering the groundwater system from septic systems farther to the north. Although farther below grade, nitrate-nitrite may still be entering into Wills Creek from this depth.

Nitrate-nitrite level, as the dissolved oxygen levels, at the study site were comparable to those encountered at the successful PRB on Cape Cod (Vallino et al. 2008). On Cape Cod the nitrate levels sat at approximately 20 mg/L, which is seen in some of the wells at the Wills Creek location. The dissolved oxygen is also low enough in most of the wells to allow for denitrification to occur if there is enough organic carbon. We might expect similar performance of a PBR at Wills Creek.

Organic carbon content in groundwater is generally a direct result of the overlaying soil content, and a raised amount of organic carbon can show contamination (Christensen et al., 1997). When manure was stored directly on the ground for 20 years, groundwater close to the storage site had organic carbon levels higher than groundwater farther away (Urbaniak and Pietrzak, 2011). High levels of carbon could also come from runoff in the shallow groundwater wells.

In order to install a permeable reactive barrier, the hydraulic conductivity of the barrier will need to exceed that of the surrounding area. Groundwater flow will follow the path of least resistance to arrive at a surface water system. If the hydraulic conductivity of the barrier exceeds that of the surrounding area, groundwater surrounding the barrier will be “pulled” into the barrier from the sides and above and below where the barrier is actually installed (Figure 15). This will allow a greater volume of water to be denitrified and therefore allow less nitrate contamination to enter the surface water system at Wills Creek and eventually the Forge River.

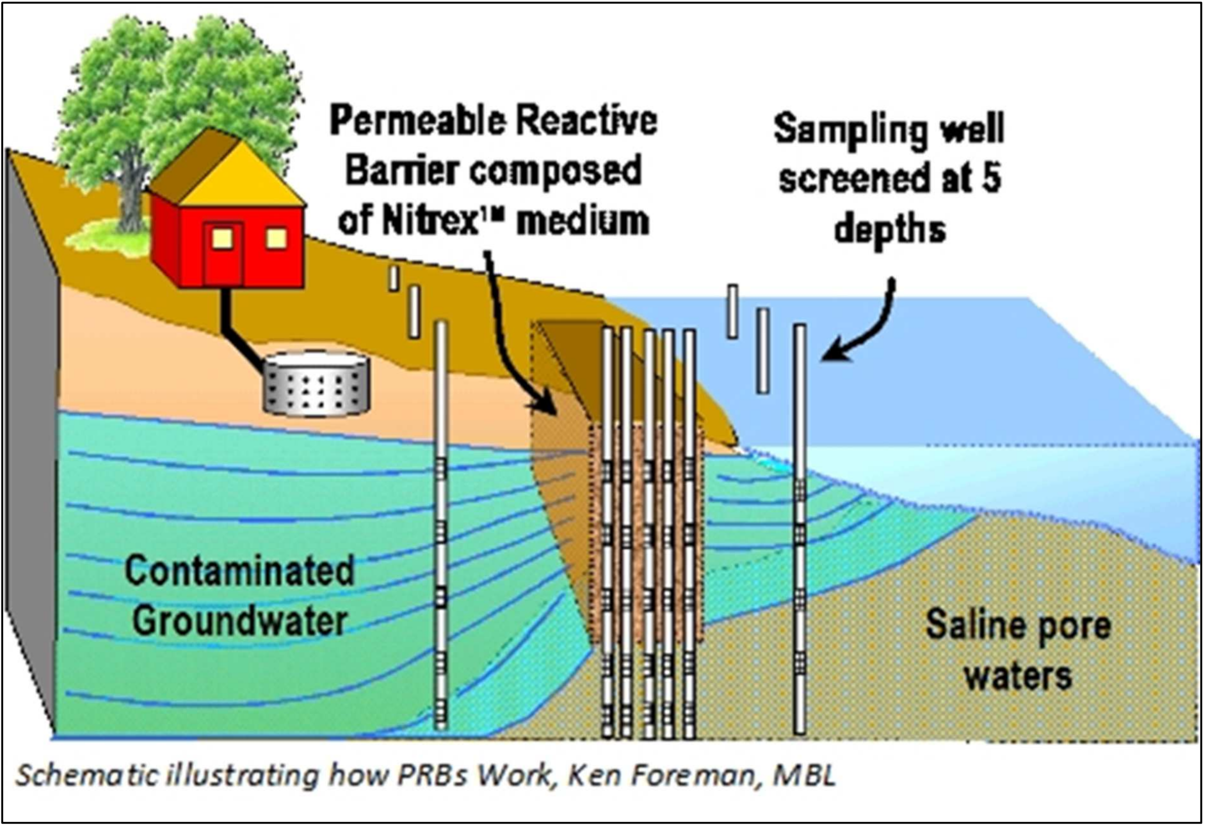


Figure 15. Hydraulic Conductivity Install. If the hydraulic conductivity in the barrier is higher than that of the surrounding area, the PRB will "pull" water surrounding the barrier into it and allow more contaminated groundwater to be denitrified. (Picture from Falmouth Water Stewards www.facesfalmouth.org accessed 2014)

Conclusion

The shoreline of Wills Creek would be an acceptable study area for a permeable reactive barrier. The monitoring wells have a level of nitrate that is comparable to the study area on Cape Cod. Also, the dissolved oxygen is low enough to allow denitrification to occur and would tend to be depressed further but reactions in the PRB itself. Water table plumes could be intercepted at shallow depths in the vicinity of MW-1 and MW-2. The PRB would need to be able to intercept deeper groundwater to reach what appears to be contamination at 4 m or 5 m below the road. Trenching below the water table can be difficult but might be done. An alternative would be to increase the hydraulic conductivity on the PRB material in order to capture groundwater flow from below. Additional measurements and modelling would be needed to investigate this latter option.

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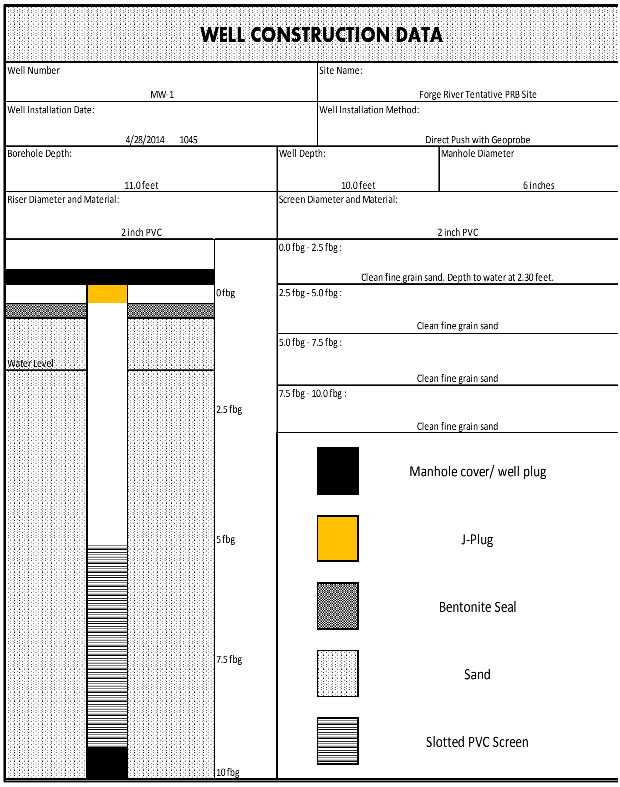
Appendices

Appendix 1: Notice to Property Owners

Monday, April 28th the Town of Brookhaven and Stony Brook University will be in the area drilling groundwater monitoring wells. There may be higher noise levels than usual between the hours of 8:30am and 4:00pm. This may also cause more traffic than usual on Riviera Drive.

Thank you for your understanding.

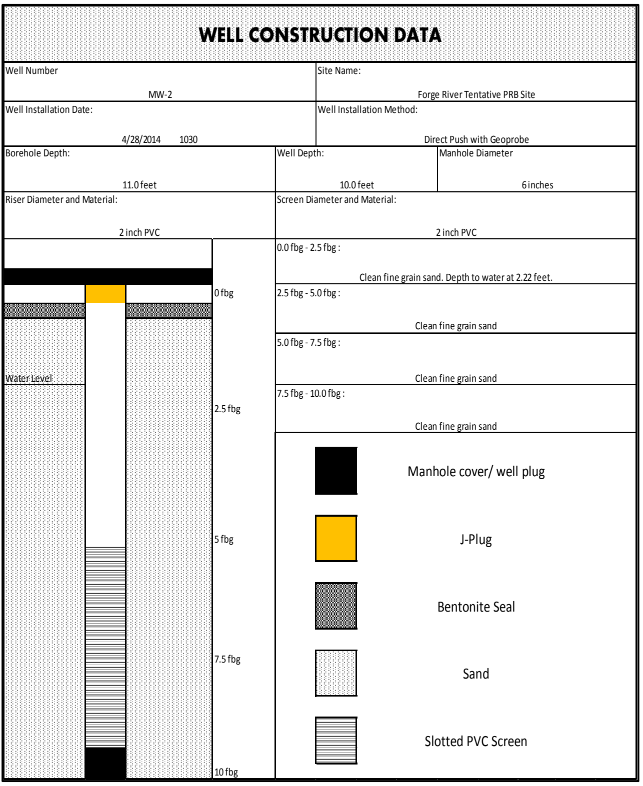
Appendix 2: Well Geometries



WELL CONSTRUCTION DATA

Well Number: MW-1
Site Name: Forge River Tentative PRB Site
Well Installation Date: 4/28/2014 1045
Well Installation Method: Direct Push with Geoprobe
Borehole Depth: 11.0 feet
Well Depth: 10.0 feet
Manhole Diameter 6 inches
Riser Diameter and Material: 2 inch PVC
Screen Diameter and Material: 2 inch PVC
0.0 fbg - 2.5 fbg : Clean fine grain sand. Depth to water at 2.30 feet.
2.5 fbg - 5.0 fbg : Clean fine grain sand
5.0 fbg - 7.5 fbg : Clean fine grain sand
7.5 fbg - 10.0 fbg : Clean fine grain sand

■ Manhole cover/ well plug
🟨 J-Plug
▩ Bentonite Seal
░ Sand
▤ Slotted PVC Screen



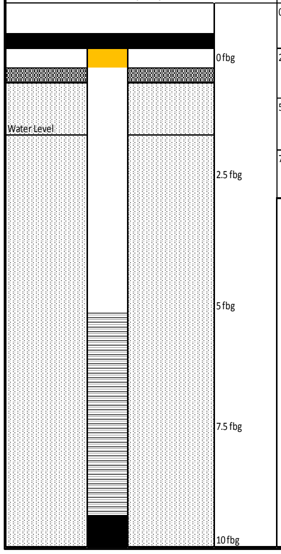
WELL CONSTRUCTION DATA

Well Number: MW-2
Site Name: Forge River Tentative PRB Site
Well Installation Date: 4/28/2014 1030
Well Installation Method: Direct Push with Geoprobe
Borehole Depth: 11.0 feet
Well Depth: 10.0 feet
Manhole Diameter 6 inches
Riser Diameter and Material: 2 inch PVC
Screen Diameter and Material: 2 inch PVC
0.0 fbg - 2.5 fbg : Clean fine grain sand. Depth to water at 2.22 feet.
2.5 fbg - 5.0 fbg : Clean fine grain sand
5.0 fbg - 7.5 fbg : Clean fine grain sand
7.5 fbg - 10.0 fbg : Clean fine grain sand

■ Manhole cover/ well plug
🟨 J-Plug
▩ Bentonite Seal
░ Sand
▤ Slotted PVC Screen

WELL CONSTRUCTION DATA

Well Number: MW-3
Site Name: Forge River Tentative PRB Site
Well Installation Date: 4/28/2014 1013
Well Installation Method: Direct Push with Geoprobe
Borehole Depth: 11.0 feet
Well Depth: 10.0 feet
Manhole Diameter 6 inches
Riser Diameter and Material: 2 inch PVC
Screen Diameter and Material: 2 inch PVC

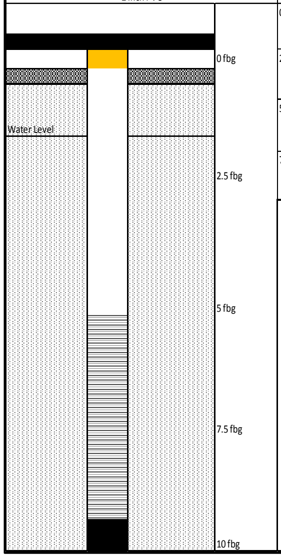


0.0 fbg - 2.5 fbg : Clean fine grain sand. Depth to water at 2.12 feet.
2.5 fbg - 5.0 fbg : Clean fine grain sand
5.0 fbg - 7.5 fbg : Clean fine grain sand
7.5 fbg - 10.0 fbg : Clean fine grain sand

- ■ Manhole cover/ well plug
- 🟨 J-Plug
- ▩ Bentonite Seal
- ░ Sand
- ▤ Slotted PVC Screen

WELL CONSTRUCTION DATA

Well Number: MW-4
Site Name: Forge River Tentative PRB Site
Well Installation Date: 4/28/2014 0945
Well Installation Method: Direct Push with Geoprobe
Borehole Depth: 11.0 feet
Well Depth: 10.0 feet
Manhole Diameter 6 inches
Riser Diameter and Material: 2 inch PVC
Screen Diameter and Material: 2 inch PVC

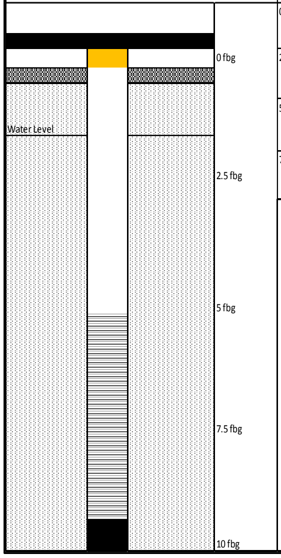


0.0 fbg - 2.5 fbg : Clean fine grain sand. Depth to water at 1.97 feet.
2.5 fbg - 5.0 fbg : Clean fine grain sand
5.0 fbg - 7.5 fbg : Clean fine grain sand
7.5 fbg - 10.0 fbg : Clean fine grain sand

- ■ Manhole cover/ well plug
- 🟨 J-Plug
- ▩ Bentonite Seal
- ░ Sand
- ▤ Slotted PVC Screen

WELL CONSTRUCTION DATA

Well Number: MW-5
Site Name: Forge River Tentative PRB Site
Well Installation Date: 4/28/2014 0925
Well Installation Method: Direct Push with Geoprobe
Borehole Depth: 11.0 feet
Well Depth: 10.0 feet
Manhole Diameter 6 inches
Riser Diameter and Material: 2 inch PVC
Screen Diameter and Material: 2 inch PVC

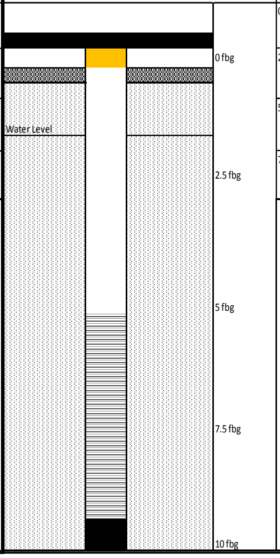


0.0 fbg - 2.5 fbg : Clean fine grain sand. Depth to water at 2.03 feet.
2.5 fbg - 5.0 fbg : Clean fine grain sand
5.0 fbg - 7.5 fbg : Clean fine grain sand
7.5 fbg - 10.0 fbg : Clean fine grain sand

- ■ Manhole cover/ well plug
- 🟨 J-Plug
- ▩ Bentonite Seal
- ░ Sand
- ▤ Slotted PVC Screen

WELL CONSTRUCTION DATA

Well Number: MW-6
Site Name: Forge River Tentative PRB Site
Well Installation Date: 4/28/2014 0900
Well Installation Method: Direct Push with Geoprobe
Borehole Depth: 11.0 feet
Well Depth: 10.0 feet
Manhole Diameter 6 inches
Riser Diameter and Material: 2 inch PVC
Screen Diameter and Material: 2 inch PVC

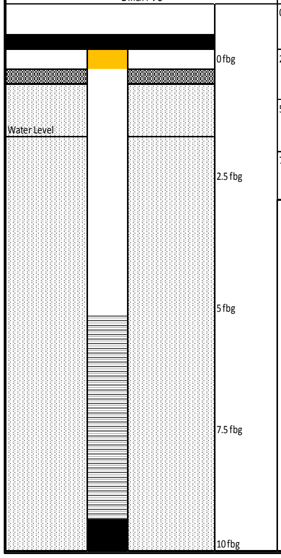


0.0 fbg - 2.5 fbg : Clean fine grain sand. Depth to water at 1.92 feet.
2.5 fbg - 5.0 fbg : Clean fine grain sand
5.0 fbg - 7.5 fbg : Clean fine grain sand
7.5 fbg - 10.0 fbg : Clean fine grain sand

- ■ Manhole cover/ well plug
- 🟨 J-Plug
- ▩ Bentonite Seal
- ░ Sand
- ▤ Slotted PVC Screen

WELL CONSTRUCTION DATA

Well Number MW-7	Site Name: Forge River Tentative PRB Site	
Well Installation Date: June 10, 2014 0900	Well Installation Method: Hollow Stem Auger with Geoprobe	
Borehole Depth: 11.0 feet	Well Depth: 10.0 feet	Manhole Diameter 6 inches
Riser Diameter and Material: 2 inch PVC	Screen Diameter and Material: 2 inch PVC	



0.0 fbg - 2.5 fbg : Clean fine grain sand. Depth to water at 1.92 feet.

2.5 fbg - 5.0 fbg : Clean fine grain sand

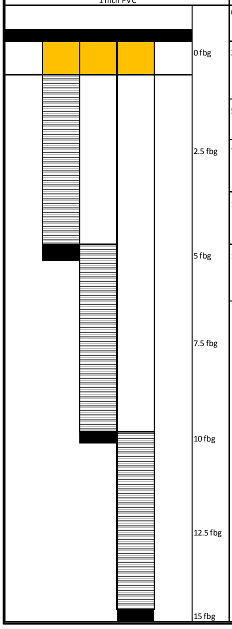
5.0 fbg - 7.5 fbg : Clean fine grain sand

7.5 fbg - 10.0 fbg : Clean fine grain sand

- ■ Manhole cover/ well plug
- 🟨 J-Plug
- ▩ Bentonite Seal
- ░ Sand
- ▤ Slotted PVC Screen

WELL CONSTRUCTION DATA

Well Number ML-1	Site Name: Forge River Tentative PRB Site	
Well Installation Date: June 10, 2014 1000	Well Installation Method: Hollow Stem Auger with Geoprobe	
Borehole Depth: 16.00 feet	Well Depths: 5.00 feet, 10.00 feet, 15.00 feet	Manhole Diameter 6 inches
Riser Diameter and Material: 1 inch PVC	Screen Diameter and Material: 1 inch PVC	



0.0 fbg - 2.5 fbg : Clean fine grain sand. Depth to water at 1.92 feet.

2.5 fbg - 5.0 fbg : Clean fine grain sand

5.0 fbg - 7.5 fbg : Clean fine grain sand

7.5 fbg - 10.0 fbg : Clean fine grain sand

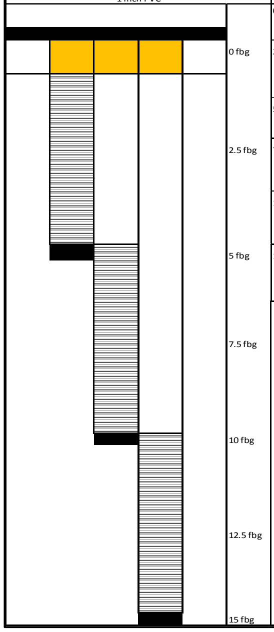
10.0 fbg - 12.5 fbg : Clean fine grain sand

12.5 fbg - 15.0 fbg : Clean fine grain sand

- ■ Manhole cover/ well plug
- 🟨 J-Plug
- ▩ Bentonite Seal
- ░ Sand
- ▤ Slotted PVC Screen

WELL CONSTRUCTION DATA

Well Number ML-2	Site Name: Forge River Tentative PRB Site	
Well Installation Date: June 10, 2014 1100	Well Installation Method: Hollow Stem Auger with Geoprobe	
Borehole Depth: 16.00 feet	Well Depths: 5.00 feet, 10.00 feet, 15.00 feet	Manhole Diameter 6 inches
Riser Diameter and Material: 1 inch PVC	Screen Diameter and Material: 1 inch PVC	



0.0 fbg - 2.5 fbg : Clean fine grain sand. Depth to water at 1.92 feet.

2.5 fbg - 5.0 fbg : Clean fine grain sand

5.0 fbg - 7.5 fbg : Clean fine grain sand

7.5 fbg - 10.0 fbg : Clean fine grain sand

10.0 fbg - 12.5 fbg : Clean fine grain sand

12.5 fbg - 15.0 fbg : Clean fine grain sand

- ■ Manhole cover/ well plug
- 🟨 J-Plug
- ▩ Bentonite Seal
- ░ Sand
- ▤ Slotted PVC Screen

Appendix 3 – Hydraulic Conductivity Calculations

Equation 3 of Bouwer and Rice (1976) can be written as:

$$K = \frac{\ln(\frac{R_e}{r_w})r_c^2}{2L}\,\frac{dln(y)}{dt} \qquad \text{[A3-1]}$$

Where ln(R_c/r_w) is given by equation 8 (Bouwer and Rice, 1976) as:

$$\ln\left(\frac{R_e}{r_w}\right) = \left[\frac{1.1}{\ln(H/r_w)} + \frac{A+B\ ln(D-(H/r_w))}{(L/r_w)}\right]^{-1} \qquad \text{[A3-2]}$$

Where D is the distance from the undisturbed water table to the base of the permeable aquifer, H is the distance between the undisturbed water table and the bottom of the screened section of the well, and A and B are empirical coefficients that had been determined in general as a function of L/r_w previously by analog modeling (Figure 3 in Bouwer and Rice, 1976).

The results from six trials in MW 4 in my study site were:

Trial	**t, seconds**	**y, Meters**	**Ln (y)**	**Ln (y)**
1	0	1.06	0.062	0.062
1	15	1.12	0.114	0.114
1	30	1.15	0.142	0.142
1	45	1.16	0.145	0.145
2	0	1.08	0.073	0.073
2	15	1.13	0.120	0.120
2	30	1.16	0.147	0.147
2	45	1.15	0.142	0.142

3	0	1.10	0.097	0.097
3	15	1.15	0.142	0.142
3	30	1.15	0.142	0.142
3	45	1.15	0.142	0.142
4	0	1.04	0.037	0.037
4	15	1.09	0.088	0.088
4	30	1.14	0.131	0.131
4	45	1.16	0.147	0.147
5	0	1.09	0.085	0.085
5	15	1.12	0.111	0.111
5	30	1.17	0.153	0.153
5	45	1.16	0.147	0.147
6	0	1.08	0.073	0.073
6	15	1.12	0.117	0.117
6	30	1.16	0.147	0.147
6	45	1.16	0.147	0.147

From a plot of ln (y) versus timd3 (Figure A3- 1] d(y)/dt = 0.0024. For the tested well (MW4) H=2.68 m, L=1.52 m, and r_w = 0.0216 m. The parameter r_c was assumed to be equal to r_w. The term ln(D-H/r_w) was given a value of 6 as advised by Bouwer and Rice (1976) in the case when D is large; based on the general geology of the south shore of Long Island, the lower boundary of the water table aquifer is likely to be the Gardiners Clay at a depth of about 30 m. Figure 3 in Bouwer and Rice, (1976), I estimated that A=3.8 and B=1.2. L/r_w = 1.52/0.0216 = 70.3

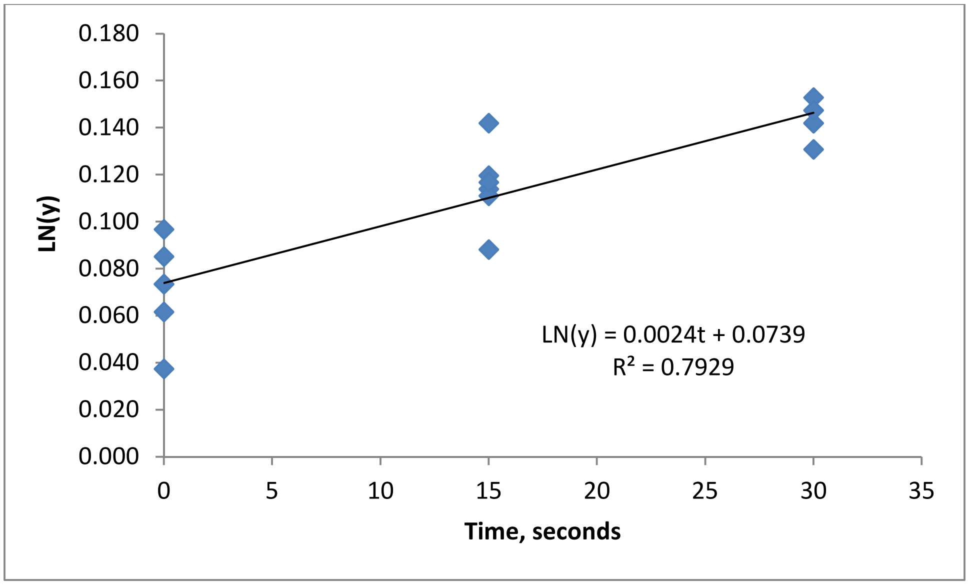


Figure A3-1.

Evaluating [A3-2]:

$$\ln\left(\frac{R_e}{r_w}\right) = \left[\frac{1.1}{4.82} + \frac{3.8+1.2*(6)}{(70.3)}\right]^{-1} = [0.376]^{-1} = 2.658$$

And, by [A3-1]

$$K = \frac{(0.0024)(0.0216)^2(2.6580)}{3.04} \text{ m/seec}$$

Or, K=0.084 m/day