

GROUNDWATER QUALITY AT A
RAPID INFILTRATION BASIN SYSTEM,
CAPE HENLOPEN STATE PARK, DELAWARE

Scott Andres, Edward Walther,
Muserref Turkmen, Changming He,
Anastasia E. M. Chirside, William Ritter

Contract Report submitted to
Delaware Department of Natural Resources
And Environmental Control

Delaware Geological Survey
2011

TABLE OF CONTENTS

INTRODUCTION

Purpose and Scope

Previous Work

 Cape Henlopen State Park

 Delmarva Groundwater

Acknowledgments

METHODS

Automated Literature Search

Field Methods for Characterization of Soils, Sediments, and Water

 Installation of monitoring devices

 Water sampling methods

 Standard monitoring wells

Laboratory methods and reporting procedures

RESULTS AND DISCUSSION

Literature Search Findings

 Contaminant transport

 N transport and denitrification

 P sorption and transport

 Effects of increased ionic strength and redox reactions

Results of Field Testing and Analyses

 Effluent quality and flow

 Impact of effluent disposal on groundwater quality-contaminant tracers

 Tracers and indicators of groundwater contamination

 Variability of tracers and N and P species with map location and time

 Variability of N and P with depth and time

 Denitrification and other redox sensitive processes

 Influence of monitoring methods and effluent disposal on observed groundwater quality

CONCLUSIONS

REFERENCES CITED

ILLUSTRATIONS

Figure

1. General location map

2. Locations of monitoring wells and surface water sampling sites
3. Results of effluent testing
4. Ternary plots of proportions of major anions
5. Ternary plot of proportions of major cations
6. Spatial distribution of specific conductance
7. Comparison of chloride concentrations and specific conductance (SC) values for surface water and groundwater and results of regression analysis
8. Spatial distribution of ammonium-nitrogen
9. Time series plot of ammonium-nitrogen for wells Ni45-43, Ni45-35, and Ni45-45
10. Spatial distribution of nitrate-nitrogen
11. Time series plots of nitrate-nitrogen and temperature for wells Ni45-43, Ni45-35, and Ni45-45
12. Comparison of specific conductance (SC) and nitrate-nitrogen concentrations for surface water and groundwater and results of regression analysis
13. Spatial distribution of orthophosphate
14. Nitrate-nitrogen – depth plots for standard-well Ni45-43 and CMT1 wells Ni45-47 through 53
15. Orthophosphate – depth plots for standard-well Ni45-43 and CMT1 wells Ni45-47 through 53
16. Nitrate-nitrogen – depth plots for standard-well Ni45-35 and CMT3 wells Ni45-61 through 67
17. Orthophosphate – depth plots for standard-well Ni45-35 and CMT3 wells Ni45-61 through 67
18. Nitrate-nitrogen – depth plots for standard-well Ni45-45 and CMT2 wells Ni45-54 through 60
19. Orthophosphate (OP) – depth plots for standard-well Ni45-45 and CMT2 wells Ni45-54 through 60.
20. Nitrate-nitrogen – depth plots for standard-well Ni45-42 and CMT4 wells Ni45-78 through 84
21. Orthophosphate – depth plots for standard-well Ni45-42 and CMT4 wells Ni45-78 through 84.
22. Comparison of dissolved oxygen and nitrate for surface water and groundwater
23. Comparison of redox potential in millivolts and nitrate in surface water and groundwater.
24. Comparison of dissolved iron and dissolved arsenic in groundwater and surface water
25. Comparison of redox potential and dissolved arsenic in groundwater and surface water
26. Comparison of dissolved oxygen and dissolved arsenic in groundwater and surface water
27. Plot of redox potential and dissolved iron in groundwater and surface water
28. Comparison of dissolved oxygen and dissolved iron in groundwater and surface water
29. Frequency distribution of arsenic concentrations aggregated by sample location
30. Comparison of orthophosphate and dissolved arsenic in groundwater and surface water

31. Comparison of specific conductance and dissolved arsenic in groundwater and surface water
32. Conceptual model of preferential flow zone

TABLES

1. Water and wastewater analysis and analytical methods
2. Results of automated search for citations containing primary key word term Rapid Infiltration
3. Results of automated search for citations containing primary key word term Soil Aquifer Treatment

APPENDICES

1. Construction and location information for monitoring wells and location information for surface water sampling sites
2. Results of water sample analyses

INTRODUCTION

Rapid Infiltration Basin Systems (RIBS) consist of several simple and relatively standard technologies employed for land-based disposal of wastewater. In Delaware, wastewater generated by parks, homes, and businesses is collected and conveyed to a treatment plant. Following processing in the treatment plant, effluent is discharged to an unlined excavated or constructed basin. By design, the effluent quickly infiltrates through the unsaturated or vadose zone to the water table portion of the Columbia aquifer.

Depending on the hydrogeologic characteristics near an individual RIBS, small quantities of effluent may slowly percolate into deeper zones of the water table aquifer or become part of a regional flow system that recharges an underlying confined aquifer.

During infiltration, some contaminants may be treated by biological and/or geochemical processes. Once in the aquifer, effluent will flow away from the infiltration site toward areas of lower hydraulic pressure, such as bodies of surface water and wells. Remaining contaminants may be treated through biologic and/or geochemical processes and diluted by dispersive and diffusive processes. The combination of contaminant attenuation and dilution processes that occur during infiltration and flow through the aquifer are termed soil-aquifer-treatment, or SAT (Crites et al., 2006).

If all of the individual components of a RIBS are working properly, there is potential for the water to be safely reclaimed for other human use and environmental purposes. This could be of benefit in Delaware, as the Columbia aquifer is a major supplier of water for potable, agricultural, and industrial users, for maintenance of surface water flow, and for sustaining important subaqueous and wetland habitat.

Purpose and Scope

This report covers one phase of a multi-year, multi-disciplinary project in which the components and the risks associated with operation of RIBS in Delaware were systematically analyzed (Figure 1). During the current phase we conducted a variety of field and laboratory experiments designed to characterize the physical, chemical, and biological controls and processes associated with the rapid infiltration of treated sewage

effluent through infiltration beds and the vadose zone. A second set of experiments was designed to characterize selected aspects of the chemical composition of geologic materials in Delaware that will likely be affected by rapid infiltration of treated sewage effluent, to characterize the geochemical effects of long-term operation of a RIBS on ground and surface waters, and to evaluate monitoring systems. Additional components of this project that are not documented in this report include an assessment of treatment plants (Turkmen et al., 2008) and an assessment of physical hydrology at an operating RIBS facility (Andres et al., 2010).

The RIBS at Cape Henlopen State Park (CHSP) was the site of a field hydrogeologic study. This site was chosen for two primary reasons. First, site access and permissions were not a problem. In this regard, a second comprehensive study at New Castle County Water Farm II was cancelled six months into the field study when the County halted all work on that facility in 2008. Second, CHSP was chosen to determine the impact of chlorinated effluent disposal from a primary treatment plant into rapid infiltration basins. Additional information on this plant is contained in Turkmen et al. (2009).

Site access and resource constraints did not allow for placement of monitoring wells in as many locations as would be needed to completely characterize the flow of effluent away from the infiltration beds and distribution of contaminants in the aquifer. For similar reasons, the analytic schedule did not include all contaminants potentially in the effluent or that could be released from the aquifer matrix due to interactions with the effluent.

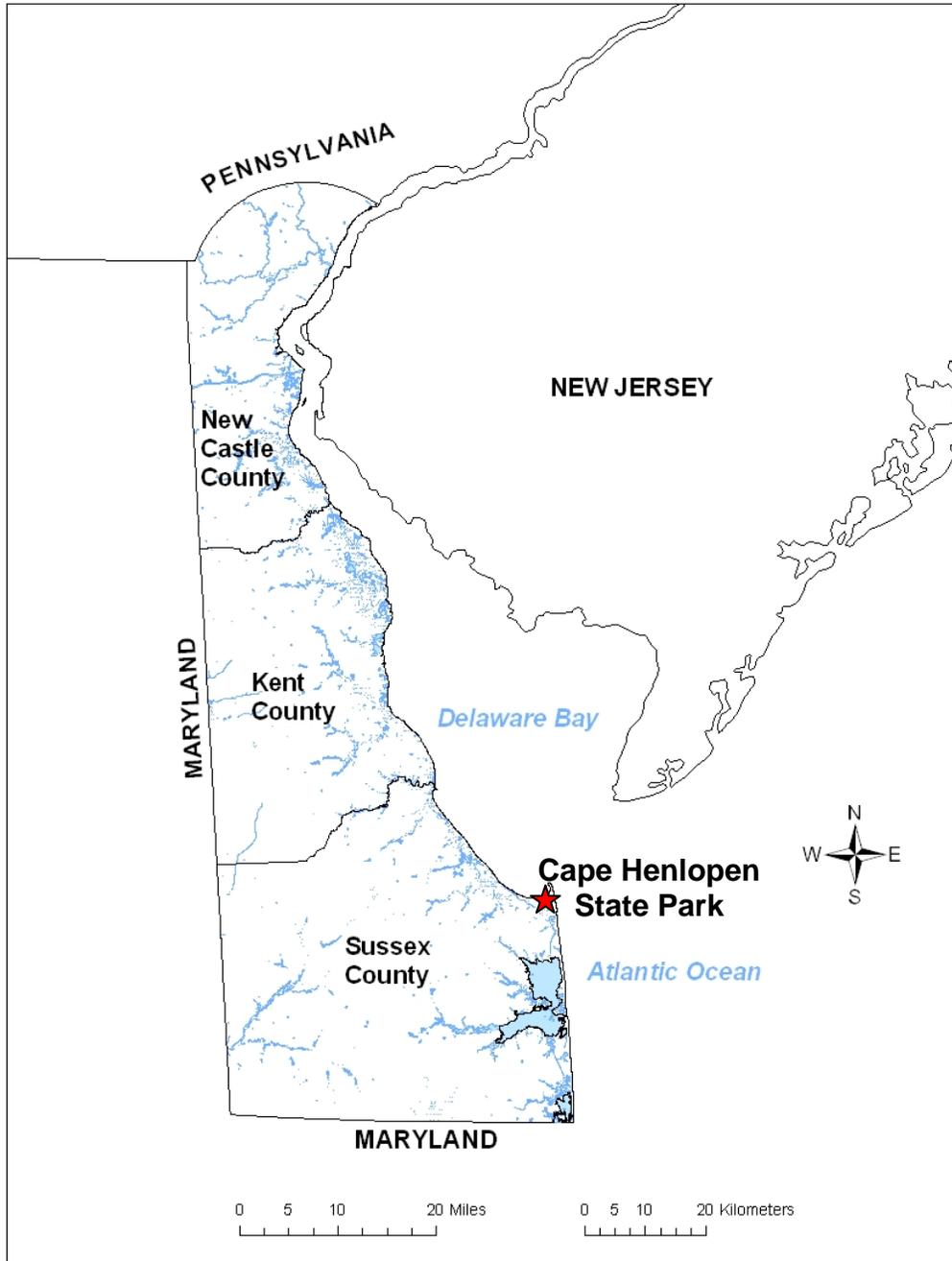


Figure 1. General location map.

Previous Work

Cape Henlopen State Park Site

The hydrogeologic framework of the Cape Henlopen State Park (CHSP) RIBS study site is characterized in Andres et al. (2010). They describe a water-table aquifer occurring within heterogeneous quartzose dune and spit complex deposits underlain by muddy marine deposits that form a leaky confining bed. Groundwater flow directions and velocities are controlled by both wastewater discharge and head difference between the higher land surface elevations associated with dune deposits and the adjacent low lying sea-level swamp (SLS, Figure 2). The SLS is a small portion of the Spit Complex Swamp described by Andres et al. (2010). Groundwater temperature and water-table configuration data indicate that flow of discharged effluent is preferentially concentrated in a zone located between the east and west edges of the infiltration basins and extending southward from the basins to the SLS. Temporal variations in groundwater temperatures due to summertime loading of warm wastewater and particle tracking models derived from water-table configurations indicate that maximum groundwater velocities are on the order of 2 to 3 feet per day between the infiltration basins and the SLS. These data also indicate a second, less well-defined flow direction oriented southeast from the infiltration basins toward the SLS. Though the water-table configuration indicates that flow should be directed towards the north, temperature data do not show the expected wastewater-caused trends, and thus flow of discharged effluent in this direction is inferred to be occurring at much smaller rates.

Military use of the area during and following World War II resulted in several potential groundwater impacts. Monitoring wells were constructed and sampled during a previous study of the area (US Army Corps of Engineers, 1997) to evaluate if there were any lasting impacts from military use. This study found no clear indication of groundwater contamination (US Army Corps of Engineers, 1997). Several of these wells were sampled as part of the current study (Appendix 1).

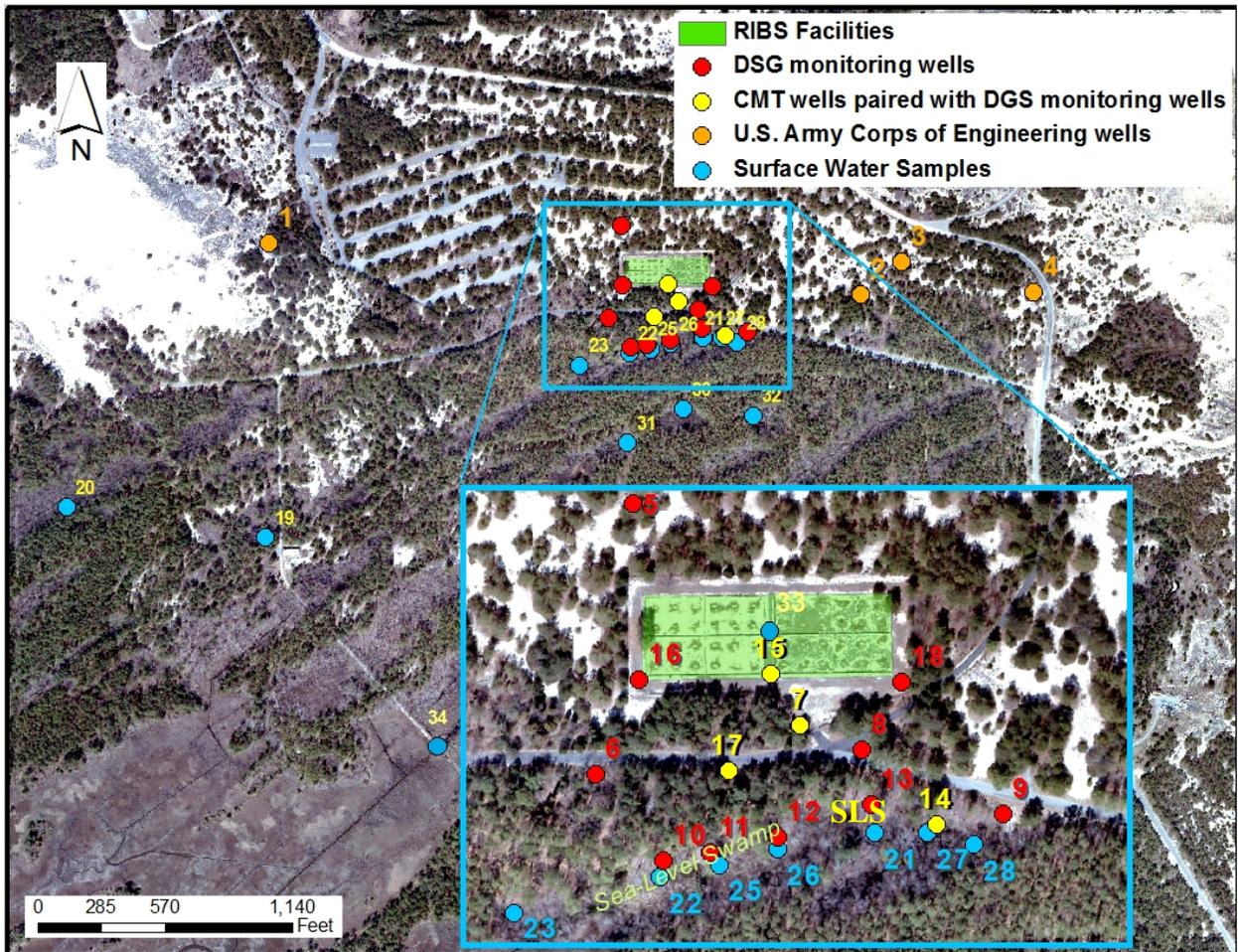


Figure 2. Locations of monitoring wells and surface-water sampling sites. CMT refers to Solinst CMT multi-port monitoring device. SLS refers to the sea-level swamp. Site identification numbers are indexed to wells listed in Appendix 1.

Delmarva Groundwater

Nearly 40 years of research by academia, state, and federal scientists have documented that the Columbia aquifer is highly susceptible to contamination from application of wastes onto and into the ground. Contaminants persist in the ground water for decades where they commonly impact potable water supply wells, and the contaminants will eventually discharge into bodies of surface water leading to well-documented eutrophication problems (Miller, 1972; Robertson, 1977; Ritter and Chirnside, 1982, 1984; Bachman, 1984; Denver, 1989; Denver et al., 2004; Andres, 1991, 1995; Center for the Inland Bays, 1995; Pellerito et al., 2006; Ferrari, 2001; Gutiérrez-Magness and Raffensperger, 2003; Debrewer et al., 2005). Ator (2008) found that Delaware water-quality problems are consistent with those observed throughout the mid-Atlantic Coastal

Plain. In this context, it should be obvious that when effluent contains nitrogen (N) or phosphorus (P) that cannot be removed or sorbed by passage through the vadose zone or aquifer, one or more of the pieces of a RIBS malfunction, or if unanticipated natural hydraulic and/or geochemical factors are present, there is substantial risk for the effluent to create conditions that will adversely impact sensitive public (water supply wells) and environmental (streams, wetlands) receptors.

There has been years of research into impacts of land-based wastewater disposal on Delmarva groundwater. Miller (1972), Robertson (1977), Ritter and Chirnside (1984), Bachman (1984), Denver (1989) have found that use of individual on-site systems (e.g., septic systems) has caused elevated concentrations of nitrate, and can lead to nitrate concentrations in excess of the drinking water standard (10 mg/L; USEPA, 2010, <http://water.epa.gov/drink/contaminants/index.cfm>). Banks and Klohe (2001) found that enteric viruses can be transported up to at least 2000 ft in the Columbia aquifer, although their study found that animal agriculture created the highest risk factor for virus transport.

Acknowledgments

This work was funded by the Delaware Department of Natural Resources and Environmental Control (DNREC) and the Delaware Geological Survey (DGS). We thank Patrick Boetcher, Joshua Kasper, and Scott Strohmeier (DNREC), Steve McCreary, Thomas McKenna, and Jaime Tomlinson (DGS), and Janine Howard, Ashley Beers, Justin Wagner, Jennalee Ruftt, Curt Romanchock, Ian McCann, and Holly Michael (University of Delaware) for their participation in the field work. The Delaware Division of Air and Waste Management provided a Geoprobe rig equipped with a single barrel coring device and operator (Patrick Boettcher) for three days. We especially thank the DNREC Division of Parks and Recreation and the staff of CHSP for permission to access property and for their assistance with logistics.

METHODS

Experimental methods were conditioned on an extensive literature search. The information sources reviewed pertained to discharge of treated wastewater onto and into the ground, biogeochemical processes affecting the fate and transport of nitrogen, phosphorus, carbon, and metals in subsurface environments, and artificial recharge of aquifers. The search included peer reviewed journal articles, technical reports from federal and state agencies, non-technical accounts in the media, and technical reports from research institutes and organizations.

Automated Literature Search

Automated literature searches were performed using the Compendex search engine, accessed through University of Delaware (UD) Library. Additional literature searches were conducted using the Google and Google Scholar applications. The Compendex search tool was especially useful for classifying and quantifying the numbers of citations according to specific topics and sub-topics from multiple journals, texts, and technical reports. Results returned by each of the two search algorithms were further evaluated to differentiate between full text, and abstracts and conference proceedings. This latter distinction separates peer reviewed from non-reviewed and potentially less reliable sources.

Abstracts of many of the documents returned from the literature search were reviewed for relevance. If abstracts were found to be relevant, then the University's on-line e-journal service was used to check for availability and to download full-text copies of journal articles. Texts were retrieved from the UD Library holdings or, in a few cases, requested with the inter-library loan service. Not all of the references checked by this process are cited in this report.

Field Methods for Characterization of Soils, Sediments, and Water

Installation of monitoring devices

Characterization of subsurface materials was accomplished through test borings and direct push methods, downhole geophysical logging, and collection and analysis of

samples as described in Andres et al. (2010). Locations and construction information for monitoring wells installed during this work and locations of surface water sampling sites are listed Appendix 1 and shown in Figure 2.

Test borings and monitoring wells were installed at nine locations (Figure 2, Sites 1-9) using the DGS CME 55 drill rig, and at five locations in the sea level swamp (SLS) and two locations near the toe of the dune using hand auger methods (Appendix 1, Figure 2, Sites 10-14). Standard monitoring wells consisting of 2-inch ID threaded flush joint schedule 40 PVC pipe and machine slotted schedule 40 PVC well screen were installed. Wells installed in holes drilled by the truck mounted equipment had 15 feet of screen. Wells installed by hand methods in the SLS and near the toe of the dune had 3 ft of machine slotted well screen. Wells installed near the toe of the dune were located at elevations a few feet higher than the SLS to help characterize groundwater that originated in the area between the infiltration basins and the SLS.

All wells were gravel packed and grouted with granular sodium bentonite. In all cases, the gravel pack extended at least 1 ft above the top of the screen interval. One to 2 ft of bentonite pellets were placed over the gravel pack. Grout was emplaced by tremie pipe in the machine-drilled holes and by hand in the hand auger holes. Protective steel casings were placed over all wells located in easily accessible areas and sealed with concrete or bentonite.

Following this work, seven-channel multi-level wells (CMT System - Solinst Canada; Einarson and Cherry, 2002) were installed next to standard wells at four locations (Appendix 1, Figure 2). Three of the CMT wells were installed with the truck-mounted drill, and one located in the SLS was installed by hand equipment. The depths of individual sample ports were determined from analysis of the descriptive and geophysical logs collected at the same site. Gravel pack was emplaced to span the interval between 2 ft above the top sample port to the bottom of the CMT tube. At three locations, 1 to 2 ft of bentonite pellets were placed over the gravel pack, and bentonite grout was then emplaced to within a few feet of land surface. Protective casings were placed over all

CMT wells and sealed with concrete or bentonite. Individual CMT channels were pumped with a peristaltic pump to remove drilling fluids and formation materials. Following completion, each channel was equipped with a dedicated sampling tube. The length of each tube was made long enough to extend from the top of the well to the bottom of the channel. The tube consists of a ¼ -inch outside diameter polyethylene tube with a stainless steel check valve on the bottom. The check valve was used to prime the tube and allow pumping with a peristaltic pump.

Water sampling methods

Water samples were collected from wells and surface water between April 2008 and November 2009 (Figure 2, Appendix 1). Sampling methods were adapted from those used in previous groundwater (Sims et al., 1996) and surface water studies (Ullman et al., 2002). Because this was a research study, no formal chain of custody procedures were employed.

During this project field measurements of pH, temperature (T), specific conductance (SC), and dissolved oxygen (DO) were made with a YSI 556 MPS. Our system included a 3-meter-long cable. This instrument permits measurements to be made within a water-tight flow-through cell or *in-situ* in a 2-inch diameter or larger well or container, or body of water greater than 6 inches in depth. The pH and SC probes were calibrated with NIST traceable standards prior to each sampling event. The DO probe was also calibrated prior to each sampling event using the manufacturer's recommended procedure (YSI, Inc., 2008).

Standard monitoring wells

Following a water-level measurement and a computation of the volume of water standing in the well, a minimum of three sample volumes were removed. For wells not located in or adjacent to the swamp, this was done with a submersible pump and the discharge tested for pH, T, SC, DO, ORP with a YSI 556 MPS meter and flow through cell. Pumping rates were always less than 1 gallon per minute (gpm) and usually less than 0.75 gpm. Turbidity was not an issue. SC, T, and pH values typically stabilized before three

sample volumes were removed. However, purging was stopped after DO and ORP readings stabilized (the rate of change of measured DO and ORP values decreased below 0.01 mg/L and 0.1 mV per 5 seconds, respectively). In some cases, DO and ORP readings did not stabilize after more than 10 well volumes were purged. When this occurred, the rate of change of DO and ORP was noted along with the final DO and ORP values. These values are recorded as estimated and less than the last measured value. A sample was collected by lowering a pre-cleaned 2.5 foot long PVC point-source bailer into the well until the top of the bailer was just below the water surface and then retrieving the bailer. This bailer kept the sample from contact with the atmosphere. The sample was pumped from the bailer with a peristaltic pump. The pump discharge was filtered through a 0.45 micron capsule filter (Envirotech) and a sample was collected in pre-cleaned bottles that were triple rinsed with the filtered pump discharge. Personnel wore fresh PVC gloves during the sample collection step. Samples were placed on ice for transport to the laboratory.

Because of difficult site access, sampling procedures for wells located in and adjacent to the swamp were slightly different. Purging was done by removing a minimum of three sample volumes from the well with a pre-cleaned bailer. Additional water was removed if the water was turbid. A sample was then removed with a second pre-cleaned PVC point-source bailer. The sample was collected using the same process as described above. The YSI 556 MPS sonde was then inserted into the well to a minimum depth of 1 foot below water level. Values and appropriate qualifiers were recorded as described above.

Bailers were completely disassembled and the parts and cord were cleaned with laboratory grade detergent and triple rinsed with deionized water between wells. To further reduce the risk of cross contamination from other wells or the cleaning process, the bailer and cord were rinsed with water discharged by the submersible pump.

Because of their different design, CMT wells were sampled using a slightly different process. A CMT well consists of 7 individual sampling channels, with each channel

having a sample port located at a different depth. Each tube was stored in a clean plastic bag at the DGS facility, brought to the site, and inserted into the sampling channel. The tube was attached to a peristaltic pump, and a minimum of 8 L of water was purged at flow rates of 200 to 400 mL/minute. Given that the capacity of the sample channels is 0.04 L per linear foot, 8 L represents 200 times the volume of standing water in the shallowest ports, and nearly 20 times the volume of standing water in the deepest sample ports.

Because of the long times needed to acquire stable measurements of DO and ORP, and early results indicated only small differences in DO and ORP between sample channels, field measurements were made on only three sample channels per CMT. Typically the samples measured were from the top, middle, and bottom CMT ports. Samples were filtered and collected as described above. Following sample collection in a given sample channel, the peristaltic pump was used to pump deionized water through the sample tube and evacuate the remaining water. Each tube was dried in the lab by purging with compressed air. The short (<15 cm) silicone tube portion of the peristaltic pump was purged with deionized water between sample channels and replaced after each sampling event.

A sub-experiment to compare water quality from standard and CMT wells was conducted. At each of the three standard wells located next to a CMT well a sample was collected with the point-source bailer from the top 2.5 feet of the water column, and a second collected from the bottom 2.5 feet of the water column. Samples were filtered and collected as described above.

Field measurements of surface water pH, T, SC, DO, and ORP were made by submerging the YSI 556 MPS sonde into the water. Samples were collected a short distance from the probe in a pre-cleaned 250 ml polyethylene sample collection container. A telescoping extension rod attachment was used if the sample collection container could not be dipped by hand without disturbing bottom sediments. Unfiltered samples were poured into pre-cleaned bottles that had been triple rinsed with the water from the sampling site. Filtered

samples were collected by pumping water from the sampling container with the peristaltic pump and filter system used for sampling monitoring wells. Care was taken not to disturb bottom sediments at any time during sampling or measurement of field parameters. Samples were stored and transported to the lab as described above.

Laboratory Methods and Reporting Procedures

Upon return to the University of Delaware, samples were stored in a refrigerator at 4 degrees C until analyzed. Samples collected in April 2009 were frozen until they were analyzed in October. Laboratory water quality testing was performed at the University of Delaware College of Agriculture and Natural Resources (CANR). Analytical procedures followed standard documented methods (Table 1). Additional analytical results of wastewater samples collected at CHSP by Envirocorp, Inc. were retrieved from the files of DNREC.

Table 1. Water and wastewater analysis and analytical methods. Quantitation limits for ICP-AES and ion chromatography results are reported in Appendix 1.

| Parameter | Method | Comments- Method detection limit |
|----------------------------|--|--|
| Biological oxygen demand | SMWW 5210B, Winkler titration | 5 day BOD – 1 mg/L |
| Total Kjeldahl nitrogen | SMWW4500C | Acid digestion – 0.2 mg/L |
| Nitrate-Nitrogen | SMWW4500, NH ₄ ⁺ B,C | Titrimetric method – 0.32 mg/L |
| Ammonia-Nitrogen | SMWW4500, NH ₄ ⁺ B,C | Titrimetric method – 0.32 mg/L |
| Ortho-Phosphorus | SMWW 4500E | Colorimetric-ascorbic acid |
| Dissolved Total-Phosphorus | SMWW 4500E | Filtration; Colorimetric-ascorbic acid; Acid digestion – 0.02 mg/L |
| Alkalinity | SMWW | Titrimetric – 1 mg/L |
| Metals | SMWW3120B | ICP-AES – metal dependent ² |
| Anions - chloride, sulfate | SMWW4100A | Ion chromatography – 0.1 mg/L |

Sources: 1. Greenberg et al., 1998; 2. Quantitation limits are tabulated in Appendix 2.

In reporting the laboratory analytic results we used the following protocols. For amounts greater than the method detection limits, we provided the value reported by the laboratory with no qualification. For amounts greater than or equal to one-half the method detection limit up to the method detection limit, we provided the value reported by the laboratory with a qualifier that the value is estimated. For amounts less than one-half the method detection limit, we reported the result as not detected at the method detection limit.

RESULTS AND DISCUSSION

Literature Search Findings

The Compendex literature search found hundreds of citations that contained the key words RIBS or SAT (Tables 2 and 3). Filtering by the secondary and tertiary search terms, which refined the search to those citations pertaining to the effects of RIBS/SAT on groundwater, returned fewer citations. It is possible that the difference between the numbers of citations returned by the primary and secondary search terms is due to inadequate key word coding of the citations. However, review of the texts of more than 100 documents returned by the primary and secondary search terms found most were focused on design, testing, and performance of wastewater systems, some of which happened to employ RIBS/SAT as a means of disposal or groundwater recharge. Of the citations identified by secondary search terms, relatively few were for English-language, peer reviewed, journal articles and agency documents that provided detailed information on systematic groundwater assessments at RIBS/SAT facilities.

A more general review of the literature identified many citations of peer-reviewed journal articles and agency documents that detailed studies that, though not focused on RIBS/SAT, contained information that was related to specific aspects of RIBS/SAT. For example, Table 3 shows thousands of citations with the key words infiltration and ground water/groundwater and hundreds of citations having the key word nitrate. The general review included the following search terms and phrases: infiltration, contaminant fate and transport, pathogens in groundwater, nitrate in groundwater, phosphate in groundwater, groundwater monitoring systems, effluent and groundwater, and denitrification. A few of these citations document groundwater contamination and remediation at wastewater

disposal facilities, including RIBS/SAT facilities. Of the several hundred citations that were reviewed over 80 are cited in this report.

Table 2. Results of automated search for citations containing primary key word term Rapid Infiltration and secondary key words hydraulics and contamination. Tertiary key words include monitoring well, nitrate, ammonia, sorption and transport.

| Rapid Infiltration | Full text & Abstract | Full text | Percentage of entries | percentage of journal articles |
|---|----------------------|-----------|-----------------------|--------------------------------|
| * All document types (1980-2009) : | 684 | | | |
| Journal articles: | 522 | | 76.3 | |
| journal articles with the term "disposal": | 19 | 7 | 2.8 | 3.6 |
| journal articles with the term "reclamation": | 7 | 2 | 1.0 | 1.3 |
| Proceedings: | 5 | | 0.7 | 1.0 |
| Book chapter: | 2 | | 0.3 | 0.4 |
| * Hydraulics | 16 | 5 | | |
| journal articles: | 13 | 5 | 1.9 | 2.5 |
| Clogging | 0 | 0 | 0.0 | 0.0 |
| Infiltration | 13 | 5 | 1.9 | 2.5 |
| Modeling | 1 | 0 | 0.1 | 0.2 |
| Groundwater | 5 | 3 | 0.7 | 1.0 |
| Monitoring well | 1 | 1 | 0.1 | 0.2 |
| * Contamination | 27 | 16 | | |
| journal articles | 18 | 12 | 2.6 | 3.4 |
| Nitrogen | 2 | 1 | 0.3 | 0.4 |
| Nitrate | 0 | 0 | 0.0 | 0.0 |
| Ammonia | 0 | 0 | 0.0 | 0.0 |
| Phosphorus | 1 | 0 | 0.1 | 0.2 |
| Sorption | 1 | 0 | 0.1 | 0.2 |
| Transport | 1 | 0 | 0.1 | 0.2 |
| Pathogens | 0 | 0 | | 0.0 |
| Disinfection | 0 | 0 | | 0.0 |
| BOD | 0 | 0 | | 0.0 |
| COD | 0 | 0 | | 0.0 |
| Suspended solids | 1 | 1 | | 0.2 |
| Groundwater | 12 | 8 | | 2.3 |

Table 2 (continued). Results of automated search for citations containing permutations of primary key word term Rapid Infiltration.

| | Full text & Abstract | percentage of entries |
|---------------------|----------------------|-----------------------|
| Infiltration | 15836 | |
| journal articles | 11846 | 75 |
| Ground water | 577 | 3.6 |
| Groundwater | 1366 | 8.6 |
| Nitrogen | 381 | 2.4 |
| Nitrate | 196 | 1.2 |
| Ammonia | 81 | 0.5 |
| Phosphorus | 156 | 1.0 |

| | Full text & Abstract | Full text | percentage of entries |
|---------------------------|----------------------|-----------|-----------------------|
| Rapid Infiltration | 684 | | |
| journal articles | 522 | | 76 |
| Ground water | 43 | 26 | 6.3 |
| Groundwater | 85 | 53 | 12 |
| Nitrogen | 17 | 7 | 2.5 |
| Nitrate | 9 | 6 | 1.3 |
| Ammonia | 2 | 0 | 0.3 |
| Phosphorus | 8 | 2 | 1.2 |

| | Full text & Abstract | Full text | percentage of entries |
|---------------------------------|----------------------|-----------|-----------------------|
| Rapid Infiltration Basin | 39 | | |
| journal articles | 27 | 16 | 69 |
| Ground water | 5 | 3 | 13 |
| Groundwater | 10 | 6 | 26 |
| Nitrogen | 2 | 0 | 5 |
| Nitrate | 0 | 0 | 0 |
| Ammonia | 0 | 0 | 0 |
| Phosphorus | 2 | 0 | 5 |

Table 3. Results of automated search for citations containing primary key word term Soil Aquifer Treatment. Secondary search terms are shown in the green boxes.

| Soil Aquifer Treatment | Full text & | Percentage | percentage |
|---|-------------|------------|--------------------------------|
| | Abstract | Full text | of entries |
| | | | percentage of journal articles |
| All document types (1980-2009) : | 339 | | |
| Journal articles: | 242 | | 71.4 |
| journal articles with the term "disposal": | 21 | 13 | 6.2 |
| journal articles with the term "reclamation": | 42 | 32 | 12.4 |
| Proceedings: | 3 | 0 | 0.9 |
| Book chapter: | 1 | 0 | 0.3 |
| | | | 0.4 |
| Hydraulics | 37 | 17 | 10.9 |
| journal articles: | 28 | 17 | 8.3 |
| Clogging | 4 | 3 | 1.2 |
| Infiltration | 6 | 3 | 1.8 |
| Modeling | 1 | 0 | 0.3 |
| Groundwater & Monitoring well | 16 | 10 | 4.7 |
| | 1 | 0 | 0.3 |
| | | | 0.4 |
| Contamination | 75 | 36 | 22.1 |
| journal articles | 53 | 36 | 15.6 |
| Nitrogen | 4 | 4 | 1.2 |
| nitrate | 3 | 3 | 0.9 |
| Ammonia | 1 | 1 | 0.3 |
| Phosphorus | 1 | 1 | 0.3 |
| sorption | 0 | 0 | 0.0 |
| Transport | 0 | 0 | 0.0 |
| Pathogens | 1 | 1 | 0.3 |
| Disinfection | 1 | 1 | 0.3 |
| BOD | 0 | 0 | 0.0 |
| COD | 1 | 1 | 0.3 |
| Suspended solids | 0 | 0 | 0.0 |
| Groundwater | 37 | 25 | 10.9 |
| | | | 15.3 |
| Ground water/Groundwater | | | |
| journal articles | 242 | | 71.4 |
| Ground water | 37 | 20 | 10.9 |
| Groundwater | 143 | 90 | 42.2 |
| Nitrogen | 36 | 20 | 10.6 |
| Phosphorus | 5 | 2 | 1.5 |

Contaminant transport

Over the past 35 years, the understanding of groundwater contaminant transport has changed significantly from one of simple plumes or bubbles of contamination shaped by advection and dispersion (Freeze and Cherry, 1979), to highly complex spatial and temporal distributions of contamination shaped by variations in contaminant input, and heterogeneities in physical, biological, and chemical characteristics of aquifers (Commission on Geosciences, Environment and Resources, 1994; Scheibe and Yabusaki, 1998; Hubbard et al., 2001; Tartakovsky et al., 2007).

Key to this change in view has been the evaluation of data from increasingly more sophisticated groundwater monitoring systems, from early systems consisting of simple long-screen wells that were sampled monthly or quarterly, to later systems consisting of spatially extensive arrays of short-screened, multi-depth devices that are intensively sampled during short-term (e.g., hourly) to moderate term (e.g., monthly to bi-monthly) injection, pumping, and natural gradient experiments (Bales et al., 1997; Day-Lewis et al., 2004). Several of these experiments have been conducted in Delaware at the Dover Air Force Base (e.g., Liu and Ball, 1999; Michalak and Kitindis, 2004). Data collected from these and other sophisticated monitoring experiments have resulted in identification of complex conceptual, physical, and mathematical models that describe aquifer heterogeneity (Scheibe and Freyberg, 1995; Hubbard et al., 2001; Day-Lewis et al., 2004) and the phenomena of contaminant transport (Tartakovsky et al., 2007; Scheibe and Yabusake, 1998).

N transport and denitrification (during SAT)

There is a significant body of literature related to the occurrence of denitrification in groundwater including many reports of successful removal of N through denitrification during SAT. There also is a body of literature related to long-term and widespread N contamination of water resources due to waste disposal practices, including SAT, and agricultural and horticultural N use practices. Many such studies conducted on the Delmarva Peninsula have documented widespread N contamination of water and the

processes that control the distribution of nitrogen contamination. Literature focused on Delmarva problems prompted further research of the literature to determine if the factors that would promote denitrification and claims of successful N removal by SAT apply to Delaware.

Nitrogen removal processes described in the literature for SAT include: 1) biologically mediated aerobic oxidation of ammonium to nitrate followed by heterotrophic denitrification under anaerobic conditions where organic carbon and nitrate serve as the electron donor and receptor, respectively (Rivett et al., 2008); 2) partial aerobic oxidation of ammonium to nitrite followed by autotrophic oxidation of ammonium with nitrite serving as the electron acceptor under anoxic conditions (Asano et al., 2007). The first process is commonly used in conventional wastewater treatment plants (Asano et al., 2007). The second process, known as ANNAMOX or nitrate reduction by ammonium, has been described and demonstrated in laboratory studies (e.g., Gable and Fox, 2003; Shah and Fox, 2005); however, there are no follow-up papers on field studies in the peer-reviewed literature. Few citations of independent confirmatory field studies are in peer-reviewed literature (Rivett et al., 2008). Nitrogen gas is the final product of both processes, though nitrous oxide may result when the first process does not proceed to completion (Asano et al., 2007), and the presence of these gases in groundwater at concentrations in excess of atmospheric is an indication that denitrification, rather than simple dispersion, is reducing nitrate concentration (Rivett et al., 2008). Tracers such as stable isotopes, bromide, and chloride are also used with mixing model methods to separate the effects of dispersion from denitrification (e.g., Smith et al., 1996).

Claims made for N removal by denitrification during SAT were evaluated for their potential application to the hydrogeologic and geochemical characteristics of Delaware soils and groundwater. Study reports that were limited to literature surveys, laboratory or simulation experiments, and reports lacking well described study methods or data to corroborate conclusions were considered to need additional scrutiny prior to accepting the conclusions presented therein. We found that many citations found in conference proceedings are not available or can be located through internet access or inter-library

loan; therefore, they were not used in this evaluation. Using these criteria, we found many (>50) abstracts from conference proceedings and full-text articles from conference proceedings. Many of the citations refer to SAT facilities in arid areas of the western United States and Israel, where the SAT facilities are paired with pumping wells that withdraw the water for agricultural purposes.

This evaluation process also was applied to EPA guidance documents (USEPA, 1985, 2006) and some books (Crites et al., 2006; Crites and Tchobanoglous, 1998; Asano et al., 2007; Johnson and Pyne, 1994) that are widely cited as examples of successful SAT facilities. We reviewed the citations used to support claims of significant N removal during field studies of SAT facilities.

Most of these documents and books we reviewed reuse information from a small number of SAT facilities. In general, the information is presented in tabular format as data; however, review of the citations for these data lead to older reports containing tables presenting the exact same information as data, with the table captions citing older reports as the source of the information. We eventually determined that the primary sources of information are conference proceedings and a series of USEPA-sponsored reports of field studies completed in the 1970s at SATs located in Landis, New Jersey (Koerner and Haws, 1979), Lake George, New York (Aulenbach, 1979), Milton, Wisconsin (Benham-Blair and Affiliates, 1979), and Hollister, California (Pound et al., 1979), and summarized in Leach et al. (1980). After considerable efforts were expended in searching scientific and news media sources and phone calls to state regulatory agencies, we were unable to locate documentation for referenced SAT facilities located in Boulder, Colorado and Calumet, Michigan. Raising additional concern about the information in the USEPA guidance documents and text books is the lack of citation of findings in USEPA (2006) Crites et al. (2006), and Crites and Tchobanoglous (1998) for detailed and highly publicized studies of SATs at the Massachusetts Military Reservation and in Long Island, New York.

Review of reports on the Landis, Lake George, and Hollister sites found that the field study methods used then would not be considered adequate or complete by today's standards for routine groundwater characterization much less the detailed methods used in contaminant transport research cited in previous and following paragraphs. In general, the monitoring well transects were designed and installed to be aligned with the presumed horizontal direction of groundwater flow. Although some multi-level well clusters were used, they generally sampled less than 10 percent of the saturated aquifer thickness. These studies used quarterly sampling for less than two years. The report authors acknowledge that the resulting head data indicate that well locations may not have been adequate to characterize the direction of flow of the contaminant plume and sampling frequency may not have been adequate to characterize the variability of contamination under the field groundwater flow conditions. Independent, conservative indicators of effluent were not employed to separate the effects of improper location of monitoring devices and dispersion-induced dilution from biological removal of nitrogen (see below). These issues raise further concerns about the validity of claims for widespread, successful N removal by SAT.

Our review did find peer reviewed journal articles and agency reports that contain well documented assessments of SAT and other land-based wastewater disposal systems. The assessments report a mixture of qualified successes and failures of these systems. Many papers documenting laboratory and field studies stress the importance of the hydraulic loading rates and composition and concentration of organic carbon in the treated effluent and/or soils for promoting denitrification. Robertson et al. (2009) summarizes several other articles detailing nitrate removal by a low-tech permeable wood mulch based reactor to treat tile-drain agricultural runoff. Burde et al. (2001) describes a low-cost plant-debris-based permeable reactive reactor system developed in Germany for wastewater treatment. Peat has been used to facilitate N removal in septic effluent for decades (e.g., Brooks et al., 1984; Shahid and Viraraghavan, 1987) and systems of this type are being used in Delaware (DNREC, NonHaz database). Many laboratory studies have investigated the effects of using alternative carbon sources or modified hydraulics

on denitrification rates (e.g., Green et al., 1994). Addition of carbon is a commonly used technique in biological nitrogen removal in sewage treatment plants (Asano et al., 2007).

Researchers from the US Geological Survey (USGS) have conducted decades of intensive monitoring and process research at the Massachusetts Military Reservation (MMR). Leblanc (1984) describes how use of a RIBS consisting of secondary treatment followed by rapid infiltration into a sand and gravel aquifer led to groundwater contamination extending thousands of feet from the infiltration basins. Repert et al. (2006) report that there has been a gradual decrease in N concentrations in groundwater once the N source has been removed, but they estimated that it will take decades for the nitrate concentrations to decrease to levels that meet drinking water standards, and longer time needed for ammonium nitrogen concentrations to decrease to levels that will not cause eutrophication problems in the downflow receiving water body. It is noteworthy that the aquifer matrix and geochemistry in the aquifer at the MMR are similar to those that exist in the Columbia aquifer.

Researchers from the USGS and USEPA have also conducted intensive studies over several years at RIBS located at the East Bay and Cedar Creek water reclamation and recharge facilities in Long Island, New York. These studies found that nitrate concentrations in groundwater were reduced from those in effluent and seemed to be adequate to reduce nitrate concentrations to values that met drinking water guidelines (Oliva, 1985; Schneider et al., 1987). They cited denitrification as the cause of the nitrate reduction but did not use any independent testing that would have proved denitrification had occurred. Despite the success in reducing nitrogen, use of the recharge facilities was discontinued when problems with excessive trihalomethane concentrations in the effluent and groundwater resulting from the need to chlorinate effluent for disinfection could not be overcome (Brisbin, 1984; Oliva, 1985; Wood, 2006).

Sumner and Bradner (1996) conducted an intensive study of the Reedy Creek RIBS located near Orlando, Florida. They report qualified success for N removal, but noted concerns regarding incomplete nitrification in the vadose zone and resulting transport of

ammonium to the water table. This work was done as part of testing a 5 mgd infiltration facility (Reedy Creek Improvement District, 2009, http://www.rcid.org/Utilities_Systems.cfm). In this case, the water is reused for irrigation of cropland, orchards, and golf courses in the Orlando area.

P sorption and transport

There are a significant number of citations related to successful removal of P during SAT. There also is a body of literature related to P contamination of water resources due to SAT or other P use and disposal practices. The significance and reliability of these claims were evaluated. In much the same way as the comments regarding nitrogen, the content of the EPA guidance documents (USEPA, 1985, 2004, 2006) and some books (Crites et al., 2006; Crites and Tchobanoglous, 1998; Asano et al., 2007; Johnson and Pyne, 1994) contain case studies in which study methods are not well described and literature reported performance data are presented without critical evaluation of the data sources. For example, Table 2-11 in USEPA (2006) lists P removal performance for several facilities and draws data from Table 3.13 in Crites et al. (2000). Table 3.13 of Crites et al. (2000) does not cite the source of the performance data.

The P attenuation model contained in the EPA guidance documents and texts (e.g., Crites et al., 2000; Crites et al., 2006, Crites and Tchobanoglous, 1998) authored by others originates from Enfield et al. (1981a, b) work that was part of the larger EPA effort summarized by Leach et al. (1980). The Enfield et al. (1981a) model predicts the reaction of orthophosphate with calcareous soils through a combination of reversible P sorption and hydroxyapatite and dicalcium phosphate dehydrate precipitation or dissolution. Application of this model to land application of wastewater predicts that P is completely attenuated by the aquifer matrix, with the time and distance for complete attenuation dependent on hydraulic load, groundwater flow, and P concentration. Enfield et al. (1981b) performed calibration and validation of the model from results of detailed laboratory column experiments and from a limited field groundwater study reported in Pound et al. (1979) and Aulenbach and Clesceri (1980). The key point for evaluating

application of this particular P attenuation model to Delaware is that Delaware soils are siliceous and not calcareous (NRCS, 2010), a result of the dominantly quartzose nature of the parent geologic materials (Ramsey, 2005, 2007; Andres and Klingbeil, 2006). As a result, the inherent assumptions in the Enfield (1981a) model of an unlimited supply of calcium in the soil matrix and sorption and mineralization on calcium rich soil particles are not consistent with Delaware soils; therefore, the model is inappropriate for use with Coastal Plain soils in Delaware.

P transport and attenuation in Delaware soils and waters have been extensively studied by research teams at the University of Delaware, Department of Plant and Soil Sciences, led by Donald Sparks and J. Thomas Sims. Key findings from many laboratory experiments on Delaware soils indicate that P attenuation processes are dominated by reversible sorption on grain coatings consisting of iron and aluminum oxides and clay minerals (Sims et al., 2002). Formation of less soluble iron- and aluminum-phosphate bearing minerals are much slower than rates of sorption or desorption (Sparks et al., 2007). Results of laboratory experiments are corroborated by field observations, in that less soluble iron-phosphate bearing minerals have been rarely detected in Delaware soils having concentrations of P in excess of predicted mineral solubility limits, and when detected, they are present only at very small concentrations (Sparks et al., 2007).

Maguire and Sims (2002) found that leaching of P from Delaware soils can occur under natural precipitation intensities even when available P sorption sites in the soils are not fully occupied. Similar research has found this to be the case for loss of P from agricultural fields through overland runoff, leading to the general conclusion that soils have a finite capacity to sorb P, with the sorption capacity correlated with concentrations of soil-test-determined iron and aluminum (Sims et al., 2002). Recent efforts are now focused on determining the appropriate metrics for determining soil phosphorus capacity, which relates P, Fe, and Al concentrations in the soil and hydrologic factors to a risk factor for P loss through overland flow or leaching (J. T. Sims, oral communication).

Key caveats to results from agronomic studies of soils are that the experiments were done under conditions representative of the agronomic soil zone and rates of water application that are representative of natural precipitation or irrigation for optimizing crop production. Most soils tested were collected from the A and B horizons (Sims et al., 2002), which although representative of the shallow vadose zone, are not directly applicable to parent geologic materials in the deeper vadose or saturated zones. Because these experiments require mixing of suspended soil particles in solution for extended periods of time, they do not appear to be representative of the conditions present during rapid infiltration through a porous media, and likely overestimate the ability of soils to sorb P.

Work has been conducted on P in Delmarva groundwater by multiple researchers. Consistent with national studies, work on the Delmarva Peninsula has found that P solubility and transport are enhanced by reducing conditions (Vadas and Sims, 1998, 1999; Sims et al., 1996, 1998; Vadas et al., 2007; Kasper and Strohmeier, 2007). In general, reducing conditions coexist with concentrations of dissolved oxygen below 2 mg/L, conditions that would be expected with sufficient concentrations of natural or wastewater supplied bioavailable dissolved organic carbon (Debrewer et al., 2005; Repert et al., 2006; Parkhurst et al., 2003). It is noted however, that reducing conditions are not a requirement for observations of high concentrations (>0.1 mg/L) of P in Delmarva groundwater (Kasper and Strohmeier, 2007; Vadas et al., 2007). USGS researchers (Debrewer et al., 2005) did note the co-occurrence of measurable P and low DO, but did not detect P in enough samples to infer the processes that enhance P transport.

Research on fate of P in land applied wastewater and in-situ groundwater remediation indicates mixed success with respect to attenuation of P through sorption to the aquifer matrix or formation of P-bearing minerals. Baker et al. (1998), Zvomuya et al. (2006), Hu et al. (2005), McCobb et al. (2009), and AFCEE (2008) report that amendments of Fe or Ca to the aquifer matrix or in solution facilitates P sorption or mineralization. Amendments of alum or alum-based water treatment residuals are also used in agricultural and wastewater treatment to enhance P sorption (Asano et al., 2007).

McCobb et al. (2002), Robertson (2003), Robertson et al. (1998), Robertson and Harman (1999), Zurawsky et al. (2004), Parkhurst et al. (2003), and Stollenwerk (2003) not only discuss P attenuation but also the re-release of P back to groundwater when geochemical conditions change, such as would be expected when native groundwater or infiltrating rainwater flushes through aquifer materials that have been impacted by wastewater.

Large production-scale experiments for SAT were performed by the USGS at the East Meadow wastewater treatment facilities (Schneider and Oaksford, 1986; Schneider et al., 1987) and at the Cedar Creek wastewater treatment facility (Brisbin et al., 1984; Oliva, 1985; Wood, 2006), both located in Long Island, New York. In more than one year of operation it appeared that the SAT facilities were successfully attenuating P in the vadose zone. Unfortunately, the SAT facilities were abandoned before long-term performance could be evaluated because disinfection could not be accomplished with chlorination, and the resultant concentrations of trihalomethanes in the effluent and in monitoring wells were in excess of drinking water standards.

Results of a recent study completed at a large RIBS facility in Orange County, Florida provide information on P attenuation during SAT as well as the concept of predicting a P storage capacity of soils beneath the infiltration basin. Moura (2009) found that P concentrations in groundwater under one of four RIBS studied are similar to those in effluent. He also found that the P storage capacity of a soil predicted with agronomic testing protocols was twice that observed during leaching tests conducted at the operational infiltration rates used in the RIBS. Moura (2009) attributes the lower P storage capacity to the rapid velocity of infiltration.

Contamination of groundwater by RIBS at the Massachusetts Military Reservation has been long known and is documented in tens of agency reports and peer-reviewed articles (LeBlanc, 1984; Walter et al., 1995; McCobb et al., 2002; Parkhurst et al., 2003; AFCEE, 2008). At this site, groundwater contamination caused by use of RIBS extends thousands of feet from the RIBS and discharges to nearby Ashumet Pond, contributing to a serious eutrophication problem in the pond. The consensus of researchers is that P input from

RIBS far exceeded aquifer capability to sorb and mineralize P, a condition exacerbated in part by dissolved organic carbon from effluent that caused reducing conditions in aquifer. A reactive barrier containing zero-valent iron has been installed along the shoreline of the pond (AFCEE, 2008). Results to date indicate that this technology can be used for P abatement (AFCEE, 2008; McCobb et al., 2009).

Effects of increased ionic strength and redox reactions

Further risk for contamination is due to release of naturally occurring or anthropogenic contaminants present in the vadose zone and aquifer caused by interactions with effluent. A long-known, simple phenomenon is the ionic strength effect (Freeze and Cherry, 1979; Hem, 1992; Bolton, 2000; Szabo et al., 2004; Sparks et al., 2007), in which mineral solubility and risk for contaminant desorption increases with increasing ionic strength or increased dissolved solids (DS) concentrations. Land application of wastewater would be expected to enhance the ionic strength effect as wastewater typically contains greater concentrations of DS than Delmarva groundwater not affected by wastewater (Ritter and Chirnside, 1984; Denver, 1989).

Release and/or transport of contaminants can be facilitated by the class of biologically mediated geochemical processes known as redox reactions. Degradation of organic carbon has been cited as a common cause of changing redox conditions (Ator, 2008; McMahon and Chapelle, 2008; Jurgens, et al., 2009; Repert et al., 2006; Herbel and Fendorf, 2006). Organic carbon is a common constituent in treated wastewater (Crites et al., 2006; Turkmen et al., 2008) and has been linked to redox related release of metals including iron, manganese, mercury, and arsenic (Oren et al., 2006; Koterba et al., 2006; Repert et al., 2006; Lee and Bennett, 1998). Transport of phosphorus in groundwater has been linked to redox and organic carbon mediated reactions (Vadas et al., 2007; McCobb et al., 2002).

Results of Field Testing and Analyses

In the following discussion of water-quality data, quantitative results are discussed in terms of wells, surface water, and effluent. All results are tabulated in Appendix 2. Well and surface-water sample locations are shown on Figure 2 and listed in Appendix 1. Note that the site index numbers shown in these locations apply only to this study and do not correspond to the site numbers contained Andres et al. (2010). Repeated reference to the location map and appendices will not be made in the following discussion.

Though not explicitly expressed in every sentence, explanations of data represent results of physical and chemical analyses that have been performed on samples; that is, groundwater samples collected from wells, grab samples collected from surface water, and grab or composite samples collected from treatment plant effluent. This convention is used to avoid long, complex, compound sentences, and improve readability. For similar reasons, we will not always mention the fact that dissolved and suspended chemical data are discussed as concentrations, and that the term value will be used to discuss results of physical measurements such as specific conductance (SC in microsiemens – μS), pH (standard pH units), redox potential (Eh in millivolts – mV), and temperature (degrees Celsius – C).

Effluent quality and flow

Andres et al. (2010) report that effluent flow rate varies by more than a factor of three over the course of the study, and the sewage collection system experiences significant groundwater infiltration when the water table is high. Highest flow rates occur during periods of peak park use and when groundwater infiltration is at a maximum (Andres et al., 2010). There are periods in which significant intra-week variability in flow is observed with weekend flows somewhat greater than mid-week flows.

Effluent quality in monthly grab samples (Figure 3) varies with flow and season. Monthly composite samples, though fewer in number, have results that are similar to those in grab samples. Maximum concentrations of N and P species and maximum

temperature values are observed during the summer months when the number of park visitors and water use and wastewater generation are at a maximum (Figure 3a, 3b). Total N and P during summer months (30 mg/L and 5 mg/L, respectively) at CHSP are similar to N and P in effluent from primary treatment plants, while concentrations of total N and P during periods of non-peak park use are much lower (less than about 10 mg/L and 1 mg/L respectively) and not dissimilar to those reported for effluents from tertiary treatment plants (Asano et al., 2007, Table 3-13).

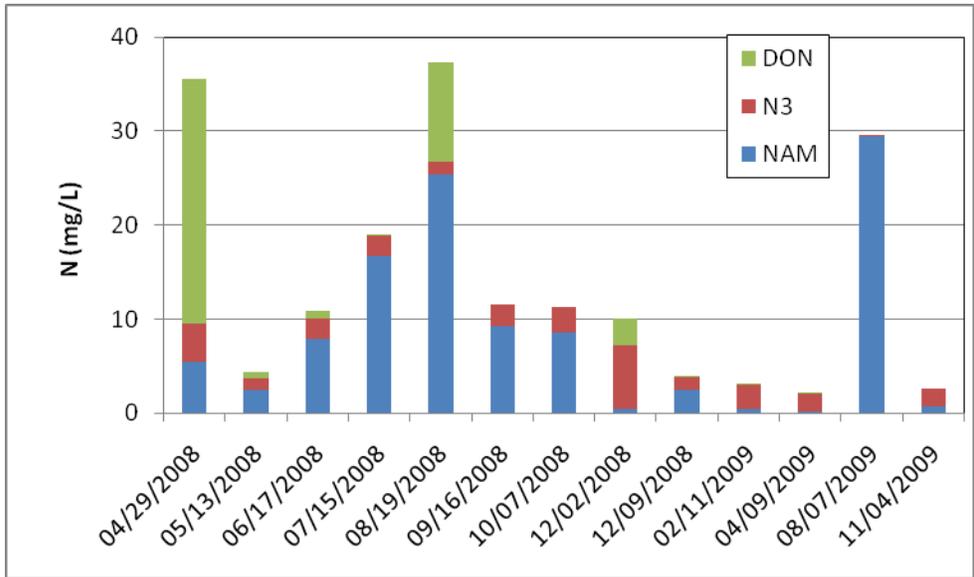
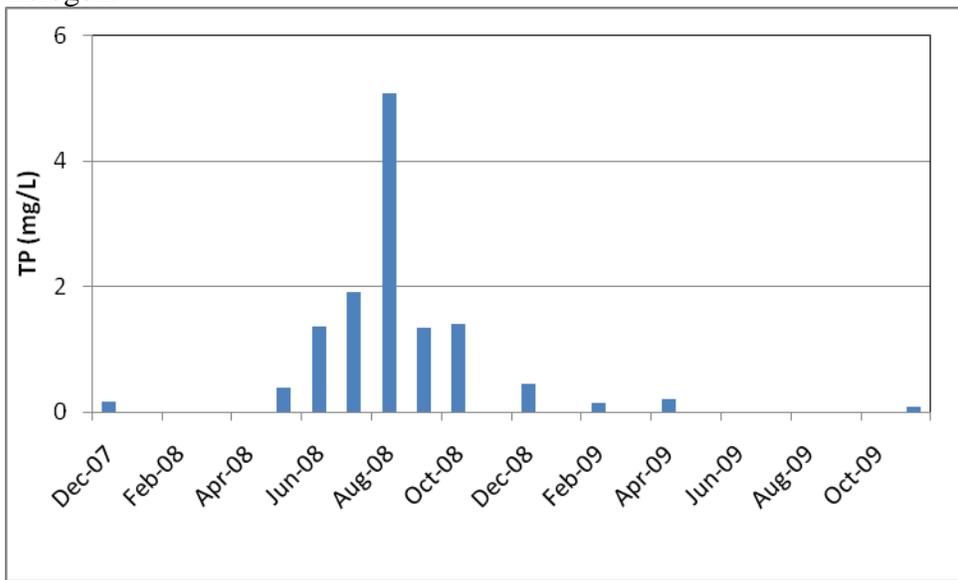
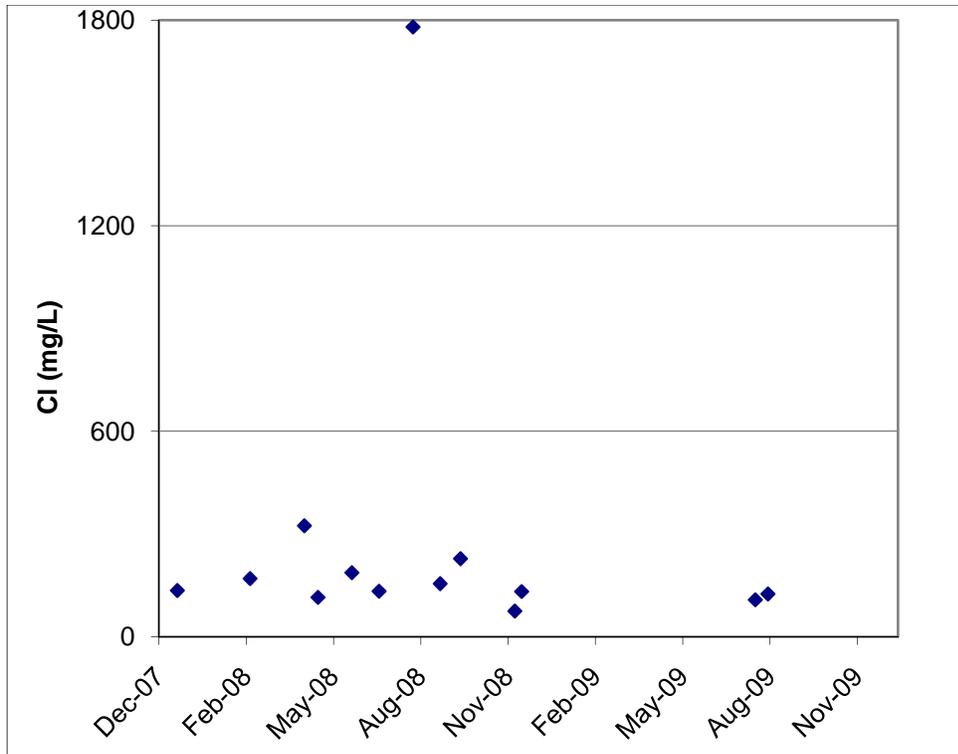


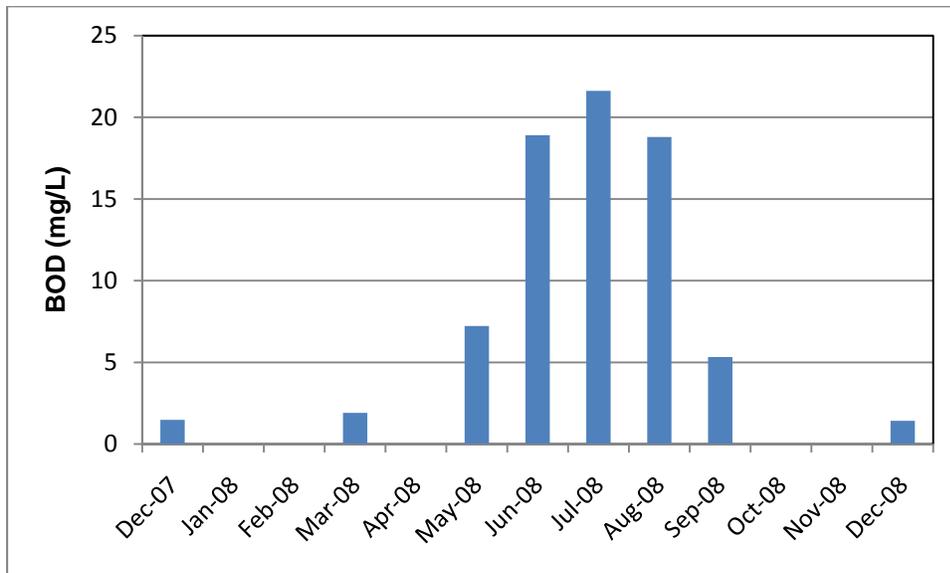
Figure 3. Results of effluent testing. Data from this study and from DNREC files. a. Plot of concentrations of N species with time. The height of the bar represents total dissolved N. DON – dissolved organic N, N3 – nitrate nitrogen, NAM – ammonium nitrogen.



b. Plot of total P (TP) concentrations with time.



c. Plot of chloride (Cl) concentration with time.



d. Plot of Biologic Oxygen Demand (BOD) concentration with time.

The monthly variability in proportions of NO_3^- , NH_4^+ , and organic-nitrogen (ON) in effluent indicate that N processing in the treatment plant varies significantly from month to month. Intraweek and seasonal variation in water use and wastewater production and time-variant rates of groundwater infiltration into the collection system appear to have affected the forms of N entering the plant and the processing of N in the plant.

The expected covariance of SC with N and P in effluent, that is high values of SC, N, and P during summer months and low values of these constituents when fresh native groundwater infiltrates into the sewage collection system, is not consistently observed. Elevated Cl^- (>300 mg/L, Figure 3c) is also noted in some samples, though interpretation is complicated by a lack of corresponding SC data. It is possible that the use of showers by bathers at the beach bathhouse may add enough Cl^- to wastewater to cause elevated Cl^- and SC values. However, high SC values and low N and P concentrations in a sample collected in April 2009 during a high water-table period, and Cl^- concentrations in excess of 500 mg/L indicate that salty water may occasionally infiltrate into the sewage collection system. We do not have enough information to determine where and if this is occurring.

Weekly and daily sampling data are generally lacking. However, samples collected on two successive days in August 2008 indicate effluent quality varies on a daily basis. Grab and 24-hour composite samples, collected by Envirocorp, contained five to ten times higher than average concentrations of Cl^- and Na^+ . An effluent grab sample collected the next day by DGS staff had a Cl^- concentration similar to average effluent. The cause of the spike in Cl^- concentrations could not be determined though the low water table at the time of sampling and the short duration of the excursion indicates an influx of salty water at the bathhouse or at one of the dormitories. Differences in N, P, and biologic oxygen demand (BOD, Figure 3d) were also observed between these samples, though not at the same magnitude as Cl^- and Na^+ . The possibility that similar short-term variability is repeated multiple times cannot be discounted.

Six samples were analyzed by ICP-AES during this study. Of note is that arsenic was measured above the detection limit in three samples, concentrations below 0.01 mg/L in two samples, and a maximum concentration of 0.023 mg/L. State and local data (Gilbert Holt, Lewes Board of Public Works) have no record of arsenic being present above the drinking water standard 0.01 mg/L in any sample from the water source serving the park.

Impact of effluent disposal on groundwater quality – contaminant tracers

Ternary plots of major anion proportions (Figure 4) show that a majority of samples and sample locations are dominated by Cl^- and NO_3^- . Anion proportions in effluent are dominated by Cl^- , but effluent contains proportionally slightly more NO_3^- than a majority of groundwater and surface water, thus plotting slightly lower in the diagram. Only a few groundwater samples from four wells plot in areas where bicarbonate and sulfate form a larger proportion of anions.

Anion data (Figure 4 - groundwater 2 group) from wells Ni45-39 (Site 11) and Ni45-41 (Site 13) plot farthest from the main data grouping and bicarbonate, not sulfate, is the anion occurring at a greater proportion. Shells, which would add bicarbonate when dissolved, were not encountered while installing these wells. Although these wells are symbolized as being in the SLS (e.g., groundwater 2 group), it is noted that these wells are located on the slope of the dune just a few feet above the SLS and not in the swamp area of the SLS. Groundwater at these sites is thought to be a mix of water that has infiltrated through wooded areas south of the infiltration basins and an uncertain proportion of effluent.

Results from two additional wells plot outside of the main data cluster. Groundwater in Ni45-50 (Site 16) is dominated by bicarbonate, and because it is located closest to the infiltration beds, the bicarbonate is thought to represent degradation of organic material in effluent. Well Ni45-47, finished a few feet shallower at this site, also contains proportionally more bicarbonate. Ni44-16 (Site 1) is located more distant from the infiltration beds than other wells and not in the preferential flow zone. Groundwater in this well contains more sulfate than bicarbonate, perhaps indicating that oxidation of

sulfide minerals may be affecting groundwater quality more at this location than in areas more strongly affected by effluent disposal. It is also possible that leaks in the sewage collection system near this well may have introduced sulfur-rich water to the aquifer. This possibility is indicated by elevated NO_3^- concentrations ($> 1 \text{ mg/L}$) in a few samples.

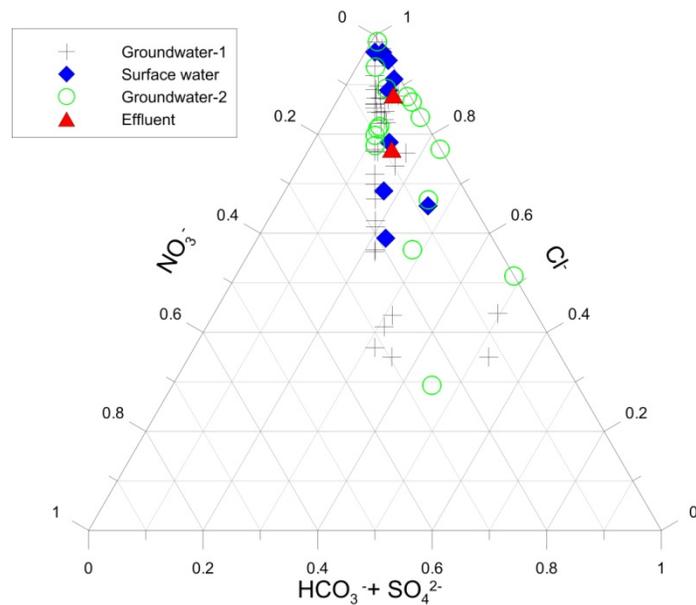


Figure 4a. Ternary plot of proportions of anions by sample. Results were averaged by sample location. Groundwater-1 indicates data from wells located at land-surface elevations greater than 10 feet NAVD88. Surface water indicates data from surface-water sources in the SLS. Groundwater-2 indicates data from wells located at land-surface elevations less than 10 ft NAVD88 in, and immediately adjacent to, the SLS. Effluent indicates data from the wastewater treatment plant.

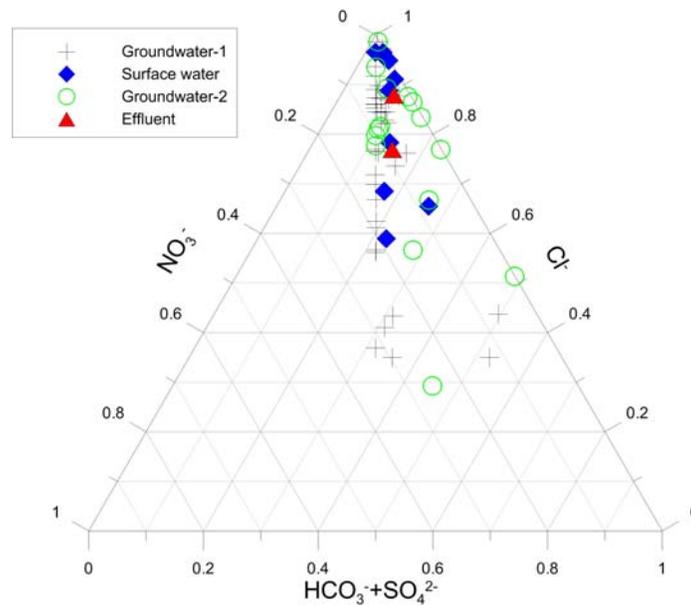


Figure 4b. Ternary plot of proportions of anions. Results were averaged by sample location. Proportions were averaged by sampling location. Groundwater-1 indicates data from wells located at land-surface elevations greater than 10 feet NAVD88. Surface water indicates data from surface-water sources in the SLS. Groundwater-2 indicates data from wells located at land- surface elevations less than 10 ft NAVD88 in, and immediately adjacent to, the SLS. Effluent indicates data from the wastewater treatment plant.

Ternary plots of major cation proportions show that a majority of samples and sample locations are dominated by Na^+ and K^+ (Figure 5). Further inspection indicates that Na^+ is the dominant cation. The dominance of Cl^- and Na^+ in major ion composition indicates that mixing of salt water into the aquifer has the greatest influence on general geochemistry at the site. An average $\text{SO}_4^{2-} : \text{Cl}^-$ ratio of 0.0278 is not diagnostic of a particular source of salt water (Hem, 1992).

Again, data from wells Ni44-16 (Site 1) and Ni45-41 (Site 13) plot outside of the main grouping of data. The increased proportions of Ca and Mg in these wells are not consistent with oxidation of sulfide minerals, and there is no direct evidence that shell bed sources of Ca and Mg are present at the sites of these wells.

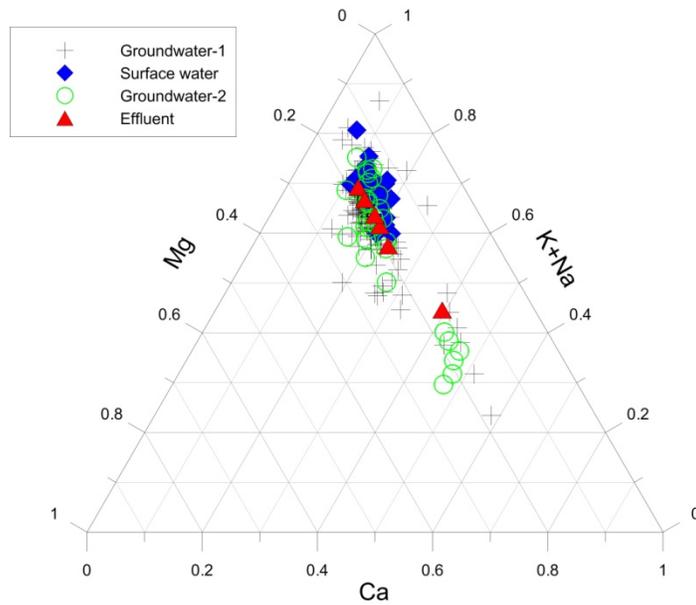


Figure 5a. Ternary plot of proportions of cations by sample. Groundwater-1 indicates data from wells located at land-surface elevations greater than 10 feet NAVD88. Surface water indicates data from surface-water sources in the SLS. Groundwater-2 indicates data from wells located at land-surface elevations less than 10 ft NAVD88 in, and immediately adjacent to, the SLS. Effluent indicates data from the wastewater treatment plant.

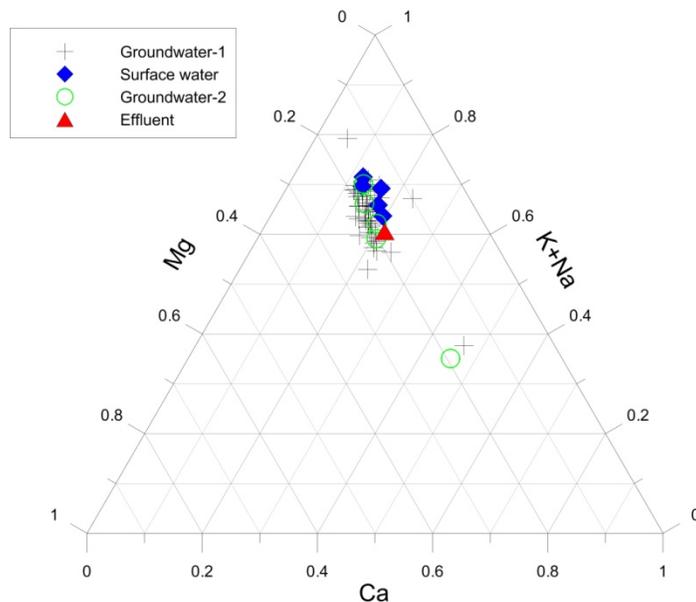


Figure 5b. Ternary plot of average proportions of cations. Proportions were averaged by sampling location. Groundwater-1 indicates data from wells located at land-surface elevations greater than 10 feet NAVD88. Surface water indicates data from surface-water sources in the SLS. Groundwater-2 indicates data from wells located at land-surface elevations less than 10 ft NAVD88 in, and immediately adjacent to, the SLS. Effluent indicates data from the wastewater treatment plant.

Tracers as indicators of groundwater contamination

Tracers of contaminant transport are physical and/or chemical constituents that are not naturally present in the aquifer, and conservative tracers are also not significantly affected by interactions with the aquifer matrix (Freeze and Cherry, 1979). Andres et al. (2010) argue that effluent temperatures during summer months are so much greater than groundwater temperatures that temperature is a useful tracer of effluent disposal in the study area. They used spatial and temporal variations in temperature to identify a preferential flow zone in which effluent from the infiltration basins flows toward the sea level swamp (SLS) located just south of the infiltration basins (Figure 2).

The choice of tracers of effluent for this study requires some discussion because of the proximity of the infiltration basins to the ocean and salt marsh and the temporal variability of wastewater quality. The chloride ion (Cl^-), a commonly used tracer, has potential for use in this study as it does not react with the aquifer matrix and is a common constituent in wastewater. Treated wastewater typically has concentrations of Cl^- , dissolved solids (DS), and SC values that are several times greater than to be expected for groundwater not impacted by wastewater (Asano et al., 2007). However, their use as tracers of effluent in groundwater in this study is somewhat complicated by variations in effluent quality. In addition, use of Cl^- as a tracer may be compromised by proximity to the ocean and the possibility of saline water in surface water and groundwater in the study area. The presence of brackish groundwater at the base of the Columbia aquifer in the CHSP has been postulated from a geophysical log in one corehole (Woodruff, 1970). Leis (1974), however, reported Cl^- concentrations less than 100 mg/L and specific conductance (SC) values less than 125 μS in surface water and shallow (< 100 ft) wells in the CHSP. This is consistent with the facts that Cl^- does not occur at significant concentrations in mineral grains that form the Columbia aquifer and groundwater from the Columbia aquifer not impacted by human activity is very dilute, containing less than 100 mg/L dissolved solids (Denver, 1989).

In practice, specific conductance (SC) is commonly used as a proxy for Cl^- and dissolved solids (Freeze and Cherry, 1979), and values less than 100 μS are thought to be

representative of groundwater not impacted by wastewater or agricultural activity (Denver, 1989). SC values less than the 125 μS reported by Leis (1974) are observed at many surface-water and groundwater sampling sites in the vicinity of the infiltration basins (Figure 6) indicating that intrusion of saline water from the ocean or other tidal water bodies should not preclude the use of Cl^- or SC as a tracer of effluent. Further indirect evidence for a lack of connection of groundwater in the study area to the ocean or salt marsh is that groundwater levels do not exhibit a diurnal tidal signal (Andres et al., 2010). There is a significant correlation between SC and Cl^- in groundwater and surface water (Figure 7).

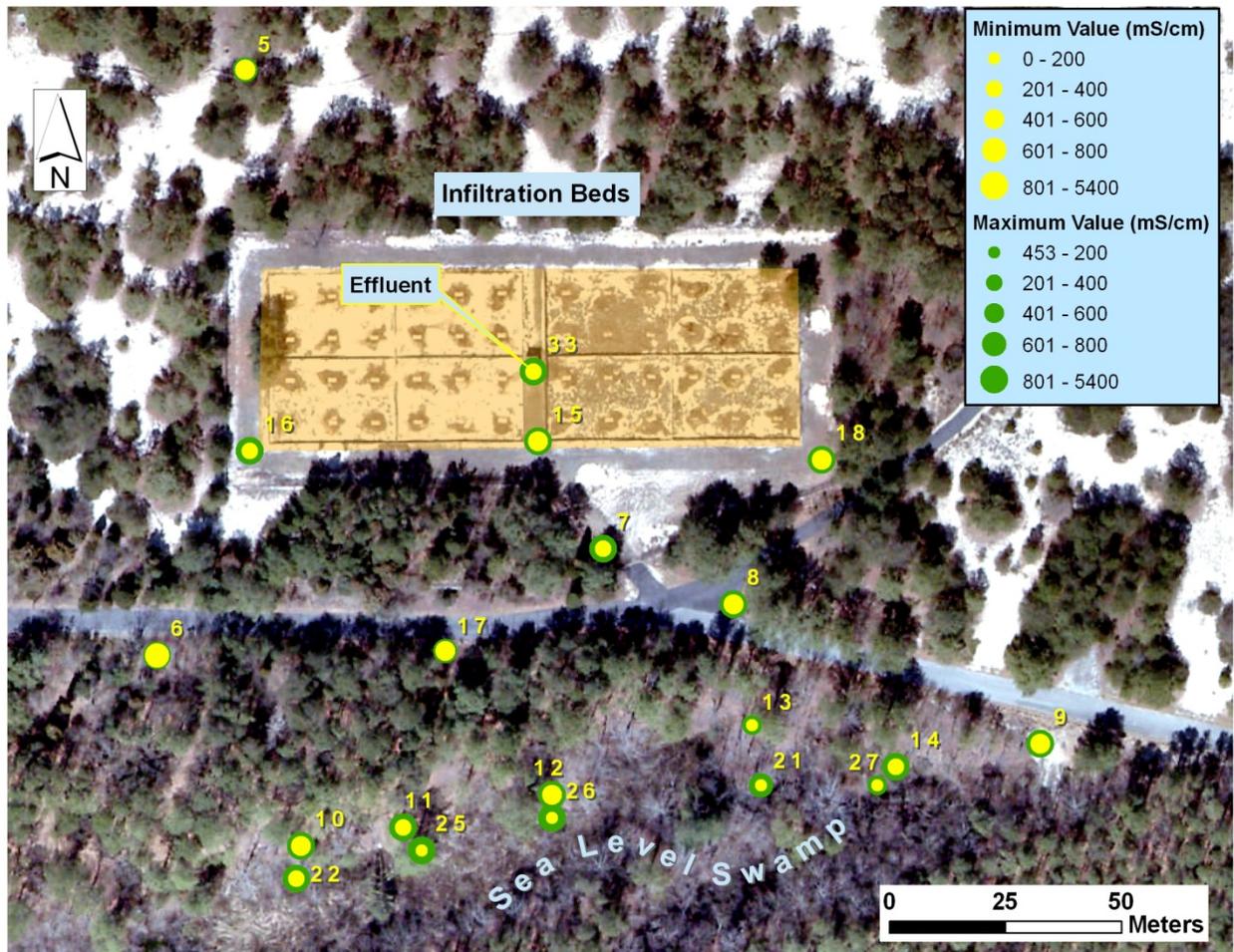


Figure 6. Spatial distribution of specific conductance. Note the generally widespread distribution of SC values greater than 600 mS/cm, thought to indicate effects of effluent disposal.

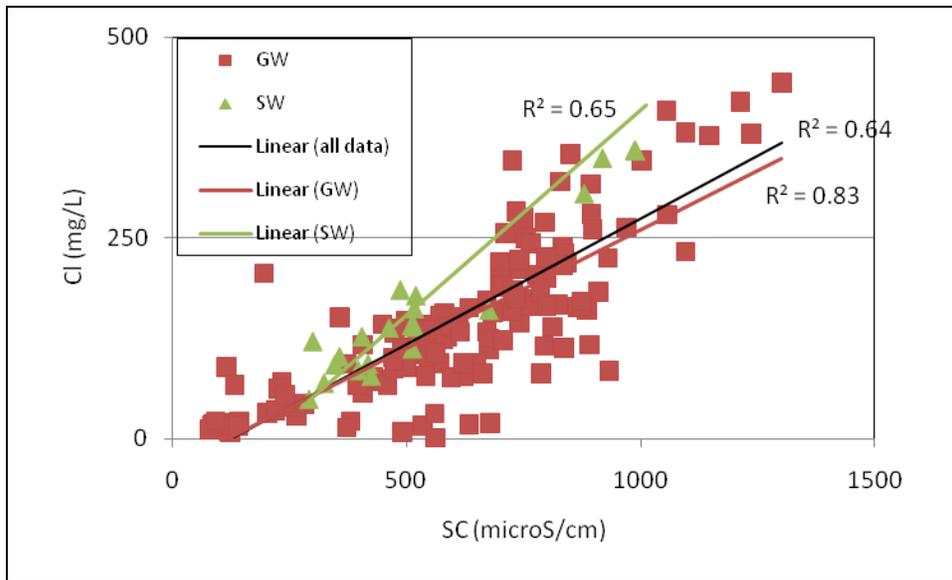


Figure 7. Comparison of chloride concentrations and specific conductance (SC) values for surface water (SW triangles) and groundwater (GW squares) and results of regression analysis. SC reported in microsiemens per cm. R-squared coefficients indicate that SC is a good predictor of Cl, and the correlations are significant ($p < 1 \times 10^{-9}$ for surface water, $p < 1 \times 10^{-34}$ for groundwater).

Variability of tracers and N and P species with map location and time

In the following discussion magnitude and variability in horizontal and temporal dimensions of SC, N, and P concentrations in surface-water, groundwater, and effluent samples are illustrated with proportionally sized and color coded symbols (Figures 6, 8, 10, 12). Lower temporal variability at a given sampling point is indicated by a small difference in size between the symbols representing minimum and maximum concentrations. Horizontal variability of concentration is indicated by differences in sizes of the symbols between sampling points. Effluent quality is schematically indicated by a single dot at Site 33 (Figure 2), rather than multiple dots located at each of the discharge pipes in each of the beds. Vertical variations of water quality are discussed in a later section.

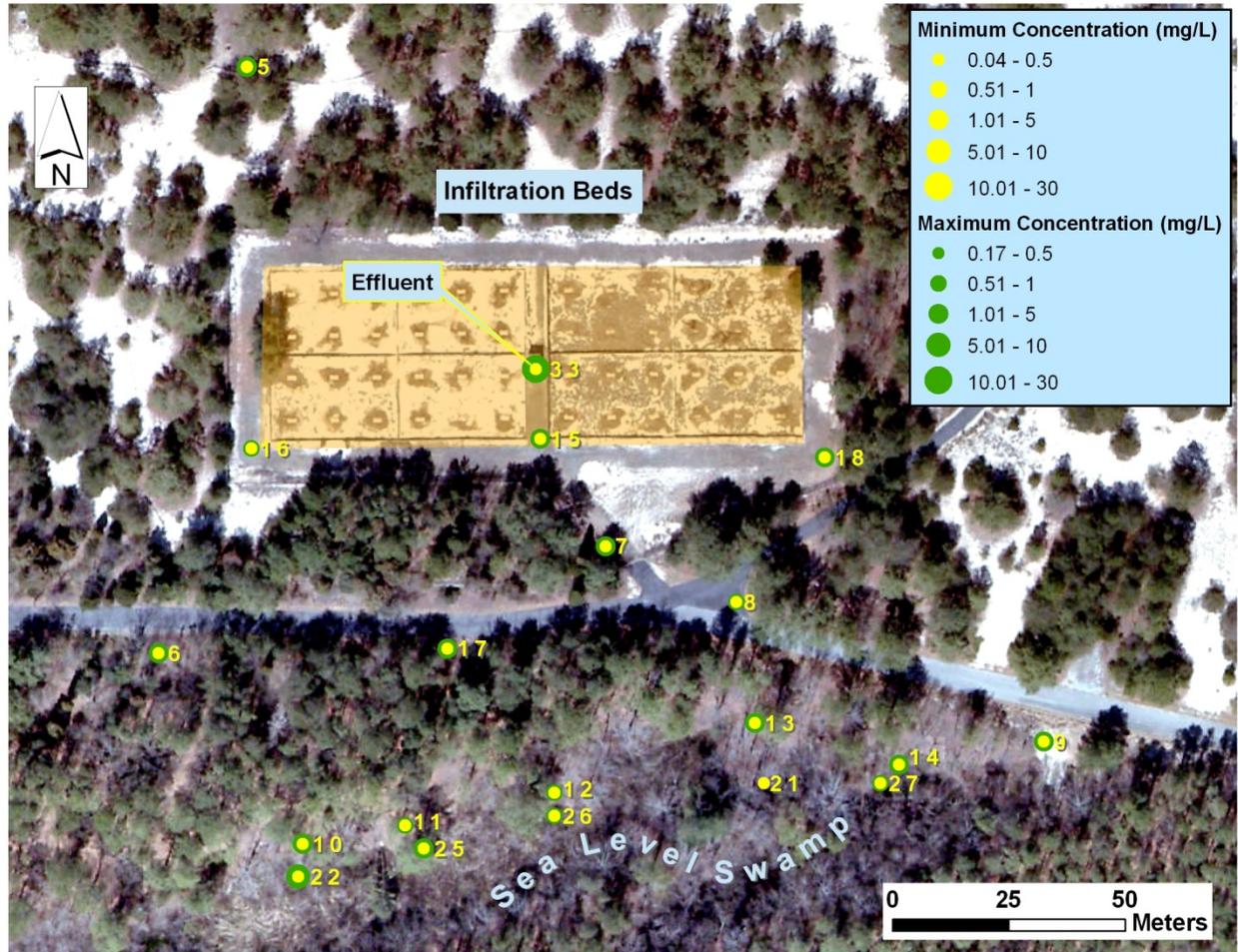


Figure 8. Spatial distribution of ammonium-nitrogen. Ammonium-nitrogen concentrations are highest in effluent samples, followed by samples from wells located closest to the infiltration basins and surface-water samples in the sea-level swamp. The difference in ammonium-nitrogen between effluent and groundwater is thought to reflect nitrification during infiltration.

SC greater than 600 μS occurs in effluent, in groundwater closest to the infiltration basins, and in groundwater and surface water within the preferential flow zone proposed by Andres et al. (2010), indicating that water quality at these sites is heavily influenced by effluent. Some of these sites show higher variability between minimum and maximum values of SC indicating that one or more factors are affecting the observed SC values. Temporally variable groundwater flow directions (e.g., Andres et al., 2010) with temporally variable effluent quality can increase differences between minimum and maximum SC in groundwater and surface water. Surface- water sites in the preferential flow zone within the SLS can be further influenced by storm runoff, which can reduce SC

through dilution. Potential impacts from contaminant sources related to the former Fort Miles (U.S. Army Corps of Engineers, 1997) cannot be dismissed entirely.

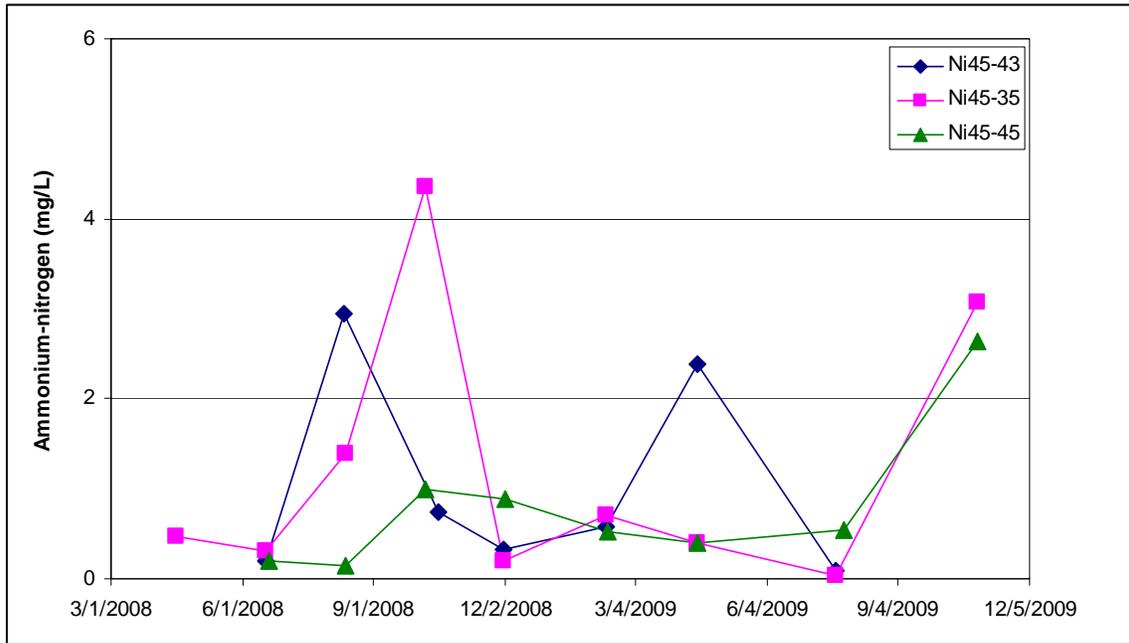


Figure 9. Time series plot of ammonium-nitrogen for wells Ni45-43, Ni45-35, and Ni45-45.

SC between 200 and 600 μS occur intermittently in groundwater (< 1 ft below water table) at sites within SLS (Sites 10-14) and at many surface-water sites (Sites 21, 25-27) (Figure 2). It is thought that water at these locations is intermittently influenced by effluent. SC between 200 and 600 μS occurs in groundwater over multiple periods Ni45-33 (Site 5) located to the north of the infiltration basins. Groundwater temperature variations were less than 1.5 degree C at this site (Andres et al., 2010) indicating the magnitude of flow from the basins to this site is occurring at a much slower rate than in the preferential flow zone.

Maximum NH_4^+ concentrations in groundwater are nearly a factor of 10 smaller than those observed in effluent indicating that NH_4^+ in effluent, or NH_4^+ produced by mineralization of ON, is nitrified in the vadose zone. NH_4^+ concentrations in groundwater are highly variable both spatially and temporally (Figure 8). The greatest concentrations and temporal variability of NH_4^+ are observed in the preferential flow

zone with peak concentrations showing later arrival times with increasing distance from the infiltration basins (Figure 9). Because the expected decrease in the magnitude of peak concentrations with distance is not consistently observed, it is likely that the wells in Figure 9 are not on the same flow path or sampling frequency was not adequate to catch the peak concentration at the well nearest to the infiltration basins. Concentrations of NH_4^+ in surface water and groundwater in the SLS tend to be slightly greater and temporally variable towards the west (Figure 8). Because this is the direction of groundwater flow within the SLS (Andres et al., 2010), this trend of higher NH_4^+ concentrations toward the west is thought to indicate greater impact of effluent in this area.

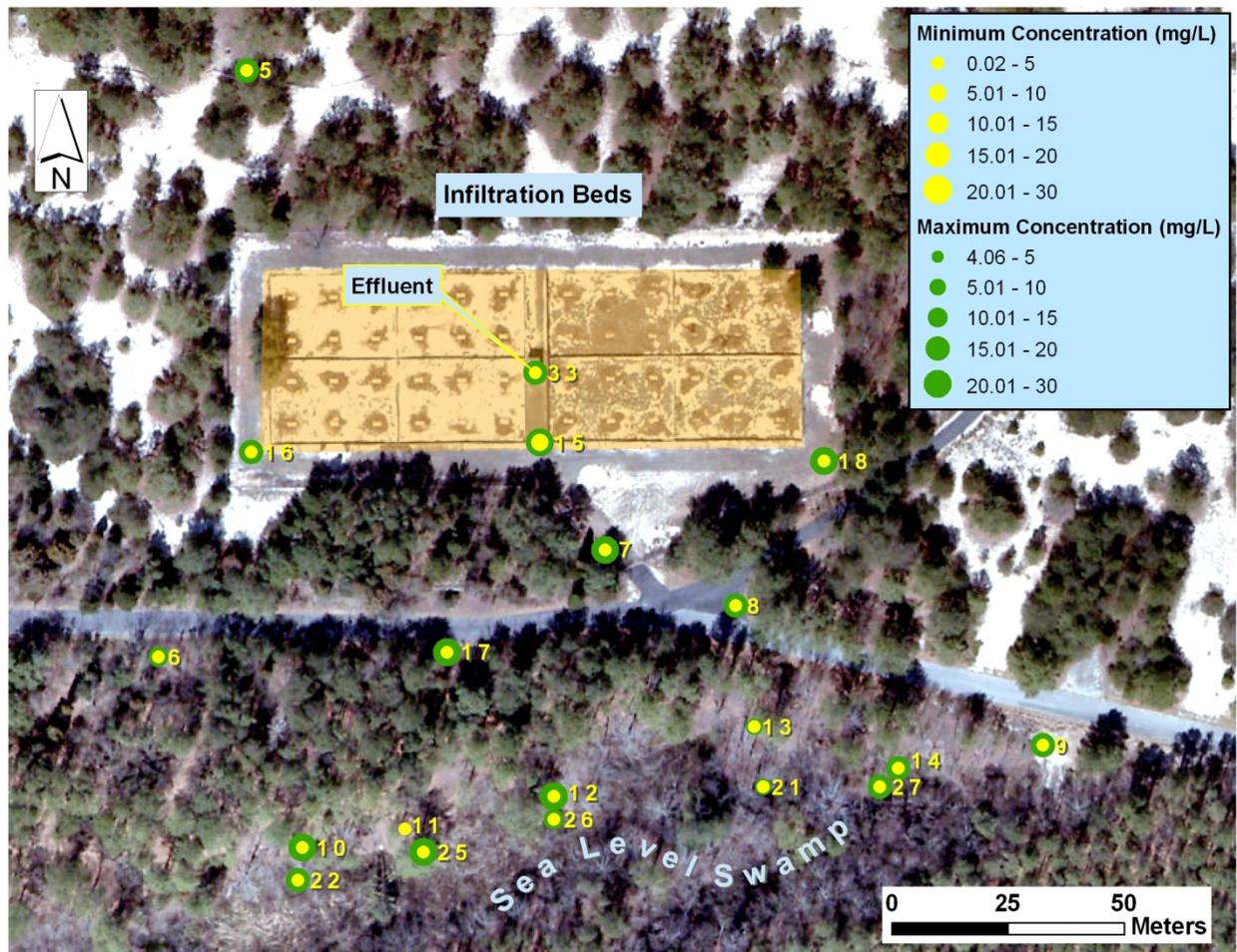


Figure 10. Spatial distribution of nitrate-nitrogen. Concentrations of nitrate-nitrogen in groundwater samples from many wells are similar to total nitrogen in effluent samples. This is thought to reflect conversion of organic and ammonium-nitrogen to nitrate-nitrogen during infiltration.

NO_3^- concentrations in groundwater are highly variable both spatially and temporally (Figure 10) ranging from non-detectable to more than 20 mg/L. Seasonal variability of N concentrations and forms of N in effluent and dilution and dispersion of effluent with distance from the infiltration basins are thought to be the dominant processes causing variability. Denitrification is discussed in a later section. The highest NO_3^- concentrations are observed in the preferential flow zone with NO_3^- -N concentrations in excess of 20 mg/L observed in wells located more than 150 ft from the infiltration basins. Time series plots of NO_3^- and temperature in three wells located at increasing distance from the infiltration beds follow similar temporal trends as NH_4^+ indicating that thermal and nitrogen signals of wastewater are transported with groundwater at roughly similar velocities (Figure 11). Similarly to NH_4^+ , the expected decrease in the magnitude of peak NO_3^- concentrations with distance is not consistently observed and it is likely that the wells in Figure 11 are not on the same flow path or sampling frequency was not adequate to catch the peak concentration at the well nearest to the infiltration basins.

Concentrations of NO_3^- -N less than 0.4 mg/L are thought to represent natural groundwater, and concentrations greater than 3 mg/L reflect human impact (Ator, 2008). Concentrations less than 0.4 mg/L are observed at wells Ni45-16 (Site 2) and Ni45-17 (Site 3), located in an area where groundwater flow paths would not transport effluent from the infiltration basins. NO_3^- -N concentrations in well Ni44-16 (Site 1), also thought to be located in an area where groundwater flow paths originating from the infiltration basins would not impact groundwater, ranged from about 0.5 mg/L to nearly 21 mg/L. These high concentrations of NO_3^- indicate another potential source of NO_3^- near this well - either the result of historic solid waste disposal, or perhaps a leaking pipe in the campground sewage collection system. Effluent disposal appears to have an impact on surface water as NO_3^- -N concentrations in excess of 5 mg/L were observed in spring 2008, prior to summer drawdown of the water table.

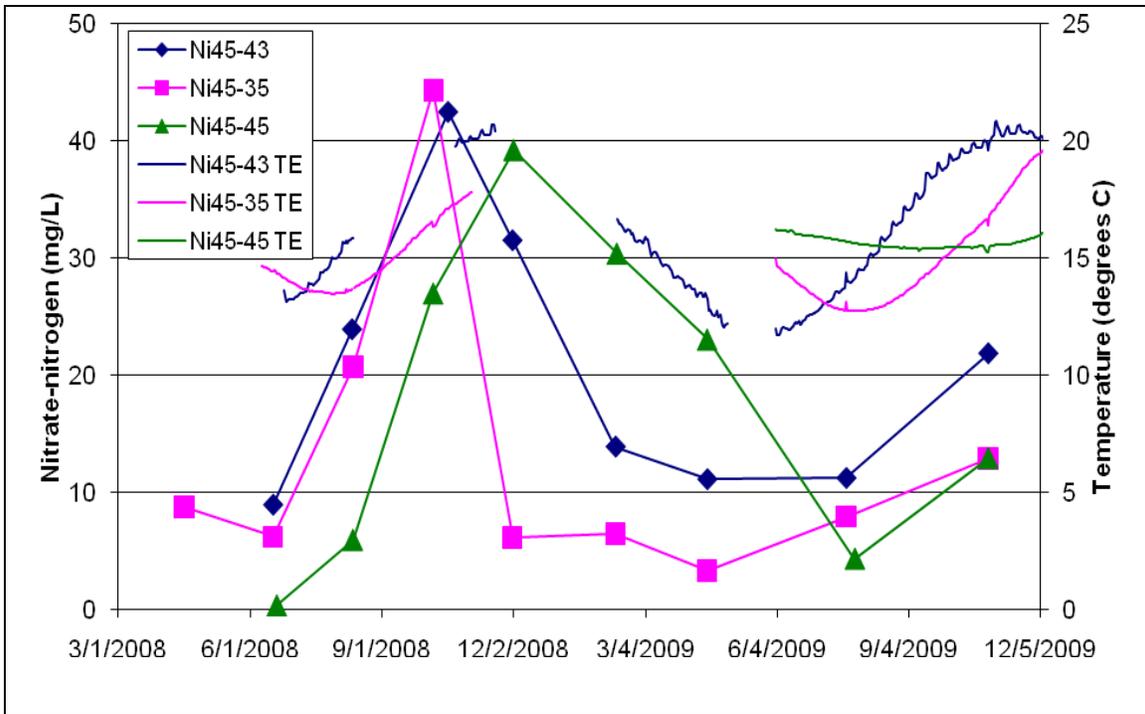


Figure 11. Time series plots of nitrate-nitrogen and temperature for wells Ni45-43, Ni45-35, and Ni45-45.

Although there appears to be spatial correlation of sites having SC values greater than 600 μS and NO_3^- -N greater than 10 mg/L, there is a poor linear correlation of SC with NO_3^- (Figure 12), indicating that SC is not a good predictor of NO_3^- .

In this discussion, total P (TP) is the sum of orthophosphate (OP), dissolved organic phosphorus (DOP), and particulate phosphorus (PP); and, total dissolved P (TDP) is the sum of OP and DOP. Recall that in this study, surface-water and effluent samples were not filtered before analysis; therefore, P values are reported as TP. Groundwater samples were filtered and; therefore, concentrations are reported as OP or TDP.

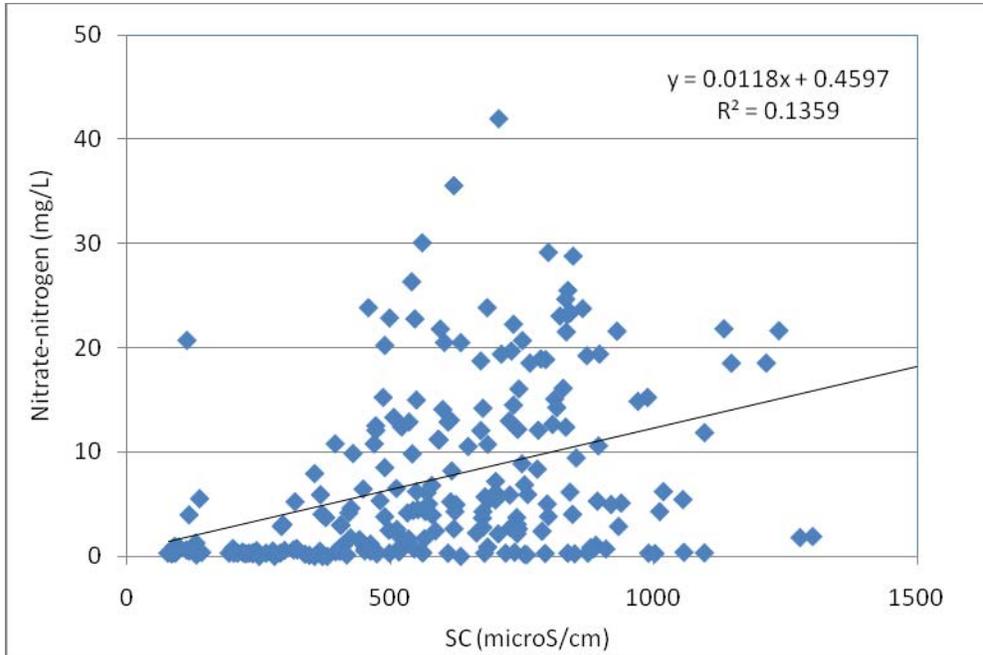


Figure 12. Comparison of specific conductance (SC) and nitrate-nitrogen concentrations for surface water and groundwater and results of regression analysis. SC reported in microsiemens per cm. The R-squared coefficient is low showing that SC is not a good predictor of nitrate. The correlation is significant at $P=1.4 \times 10^{-8}$.

OP concentrations in groundwater are highly variable both spatially and temporally (Figure 13). This primarily reflects the seasonal variability of TP concentrations and forms of P in effluent and secondly, possible effects of P sorption/desorption from the aquifer matrix. Highest OP concentrations are observed in the preferential flow zone nearest the infiltration basins, with decreasing concentrations with increasing distance from the infiltration basins. The highest OP concentrations in groundwater are slightly less than the highest TP concentrations in effluent indicating possible attenuation of P in the subsurface by physical filtering of particulate P in the subsurface or aquifer sorption effects. It is also likely that the difference between TP in effluent and OP in groundwater is an artifact of the well sampling frequency; that is, we may not have sampled wells at the time when the maximum concentrations of P were flowing through the wells.

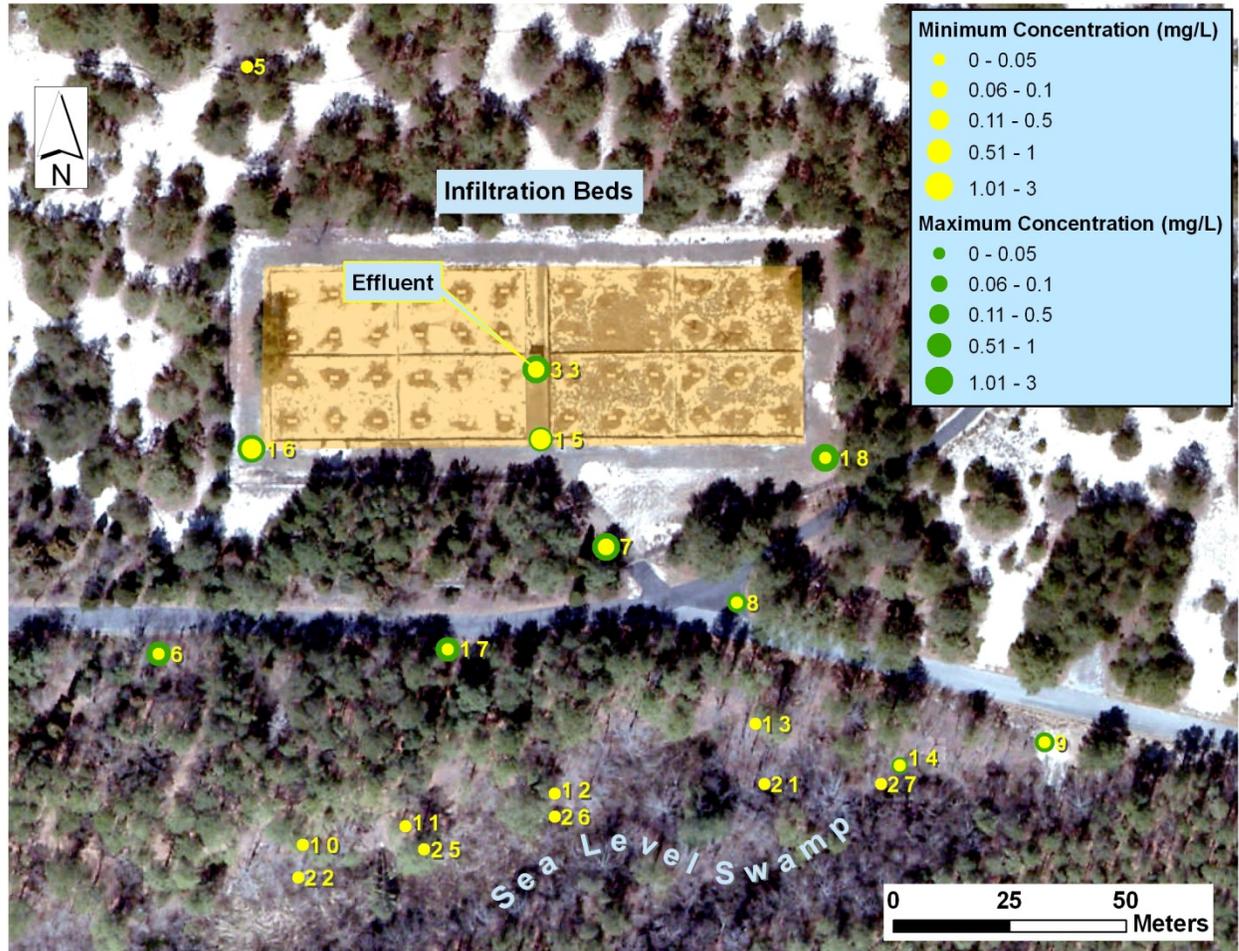


Figure 13. Spatial distribution of orthophosphate (OP). Concentrations of (OP) in groundwater samples from wells located closest to the infiltration basins are similar or slightly greater than TP in effluent samples. Note that maximum OP at site 17, located nearly 50 m from the nearest infiltration basin is greater than 1 mg/L, indicating that the aquifer matrix does not have sufficient P attenuation capacity to stop transport of relatively high concentrations of P.

The spatial pattern of differences between minimum and maximum OP concentrations is noteworthy (Figure 13). The difference between minimum and maximum OP concentration in two wells located closest to the infiltration basins is very small compared to the seasonal variability of TP concentrations in effluent. Reduced variability of OP in groundwater compared to TP in effluent is consistent with sorption of P on the aquifer matrix during periods when P in effluent is greater than P in the aquifer and subsequent desorption of P from the aquifer to groundwater during periods when P concentrations in infiltrating water are lower than those in the groundwater. Samples from several other wells in the preferential flow zone exhibit temporal variabilities of P

that are similar to that observed in effluent samples. This could indicate that groundwater is mimicking the seasonal variability of P in effluent along with lower rates of sorption and desorption from the aquifer matrix.

Concentrations of OP observed in groundwater in the preferential flow zone are much greater than the average OP concentrations of groundwater (< 0.02 mg/L) reported by Kasper and Strohmeier (2007), and greater than OP concentrations observed under fields amended with poultry litter (Sims et al., 1996, 1998).

Variability of N and P with depth and time

As discussed in the previous sections, temporal variability in effluent quality is reflected by temporal variability in groundwater quality. In addition, data from CMTs as well as data from shallow and deep samples within standard wells document that concentrations vary in a vertical direction. As a result, standard parametric tests (e.g., t-test of means, and F-tests of variance) that compare data between the group of four standard wells to the group of four CMT wells indicate that all data are drawn from the same population. The lack of discriminant power by pair-wise parametric tests indicates that these simple pair-wise tests are inappropriate for analysis of data influenced by a complex set of flow and transport processes.

Figures 14 through 21 provide comparative displays NO_3^- and OP data collected from standard wells and CMTs. These plots allow visual comparison of variations of concentrations between standard and CMT wells, variations with depth, and variations between sampling periods. Concentration scales vary by well and between sample periods to allow for vertical patterns to be seen. This display also assists discussion of groundwater quality in terms of effluent loading and flow and contaminant transport processes. For discussion purposes, CMT ports will be labeled from one to seven in order of increasing depth.

Wells Ni45-43 and CMT1 (Ni45-47 through Ni45-53) are located between two infiltration beds (Site 15) with the nearest discharge pipes located about 7 m from the

well heads. Well Ni45-43 exhibits the largest magnitude, daily effluent-related water-level and temperature fluctuations of all monitoring wells, and temporal patterns indicate that the magnitude of water-level fluctuations is related to the location of effluent discharge.

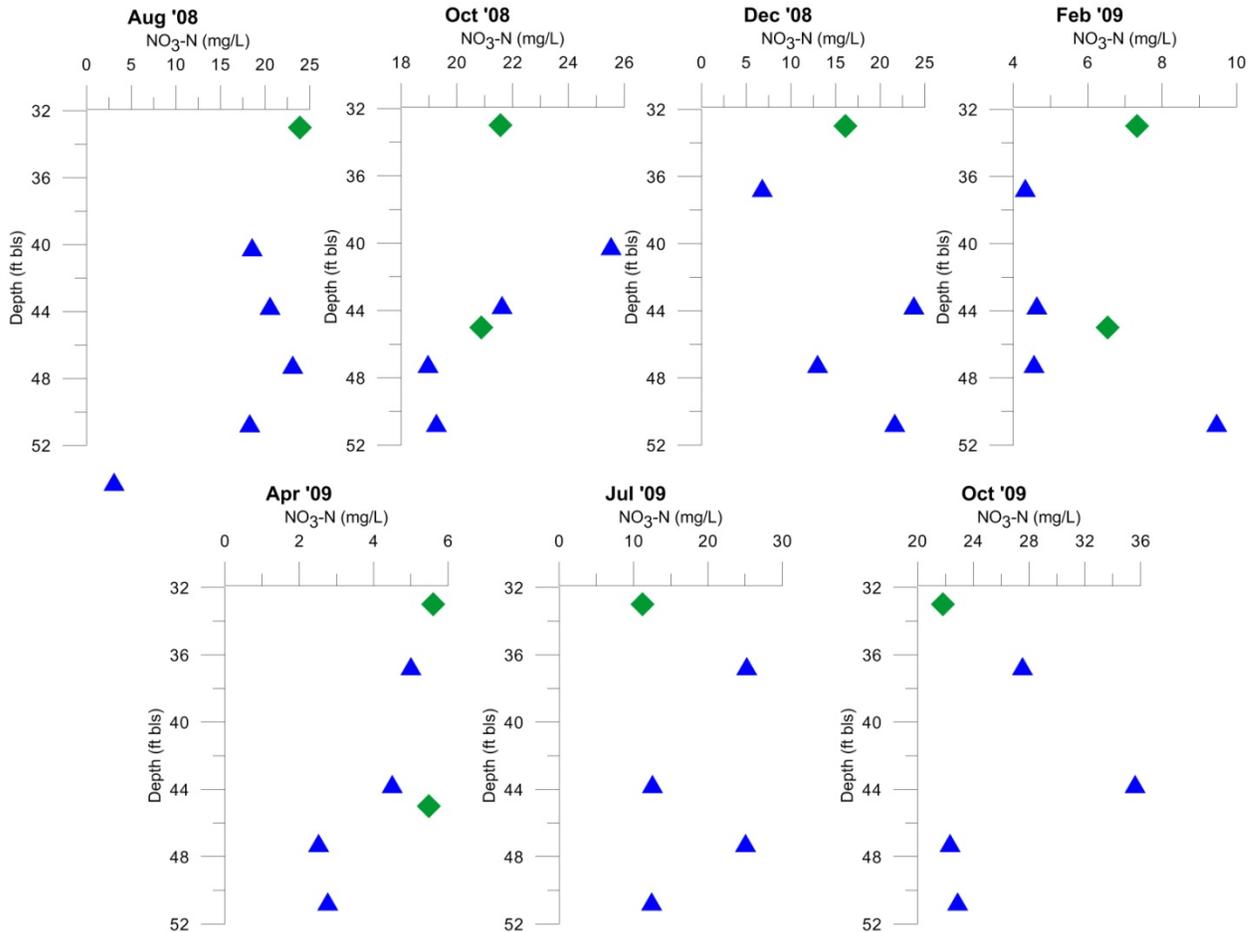


Figure 14. Nitrate-nitrogen – depth plots for standard-well Ni45-43 (diamond) and CMT1 wells Ni45-47 through 53 (triangle). Horizontal separation of wells is less than 2 m. On selected dates, the standard well was sampled at two depths with a point-source bailer.

In general, the highest concentrations of NO_3^- are observed in samples from shallowest to mid-depths in wells Ni45-43 and CMT1 (Figure 14). This is consistent with N entering the aquifer with infiltrating effluent and also indicates that effluent displaces native groundwater. Lower concentrations in the bottom sampling ports of the CMT indicate that effluent does not displace groundwater through the entire thickness of the aquifer or is possibly the result of denitrification. Denitrification is discussed in a later section.

There is significant variability in NO_3^- concentrations between sample periods, and vertical distribution of NO_3^- changes between sample periods in wells Ni45-43 and CMT1. The lowest NO_3^- concentrations occur during months following periods when there are fewer park visitors, less water use, and less production of wastewater. The highest NO_3^- concentrations occur during months following periods when there are more park visitors, more water use, and more production of wastewater. As such, NO_3^- data are consistent with interaction of spatial and temporal variabilities of effluent loading and effluent chemistry. Correlations of NO_3^- between samples from CMT ports and samples from standard wells at similar depths are inconsistent between sample periods.

OP varies substantially with depth in Ni45-43 and CMT1 (Figure 15). In general, the highest OP concentrations are observed at the shallowest depths. This is consistent with P entering the aquifer with infiltrating effluent, as opposed to a natural source of P. Groundwater OP concentrations are consistently less than TP in effluent, but variability in groundwater OP between sample periods is smaller than variability in TP in effluent indicating that there is some geochemical processing of P during effluent infiltration and during transport in the aquifer. There are no consistent correlations of P concentrations in samples from CMT ports between all sample periods. There are some similarities between some sample periods.

Well Ni45-35 and CMT3 (Ni45-61 through Ni45-67) are located downflow of the infiltration basins (Site 7). Concentrations of NO_3^- and OP in CMT ports 3-6 are typically greater than those from ports 1, 2, and 7 (Figure 16). Similarly, samples from the top of the water column in the standard well typically have lower concentrations than samples from the bottom of the well. This observation is consistent with an interpretation that contaminants have moved down in the unconfined aquifer and water at the water table originates outside of the infiltration basins.

Temporal patterns of NO_3^- in wells Ni45-35 and CMT3 are similar to each other and to changes in effluent. As discussed previously, there is a time offset between groundwater at this site and effluent that is likely related to time of travel between basins and wells.

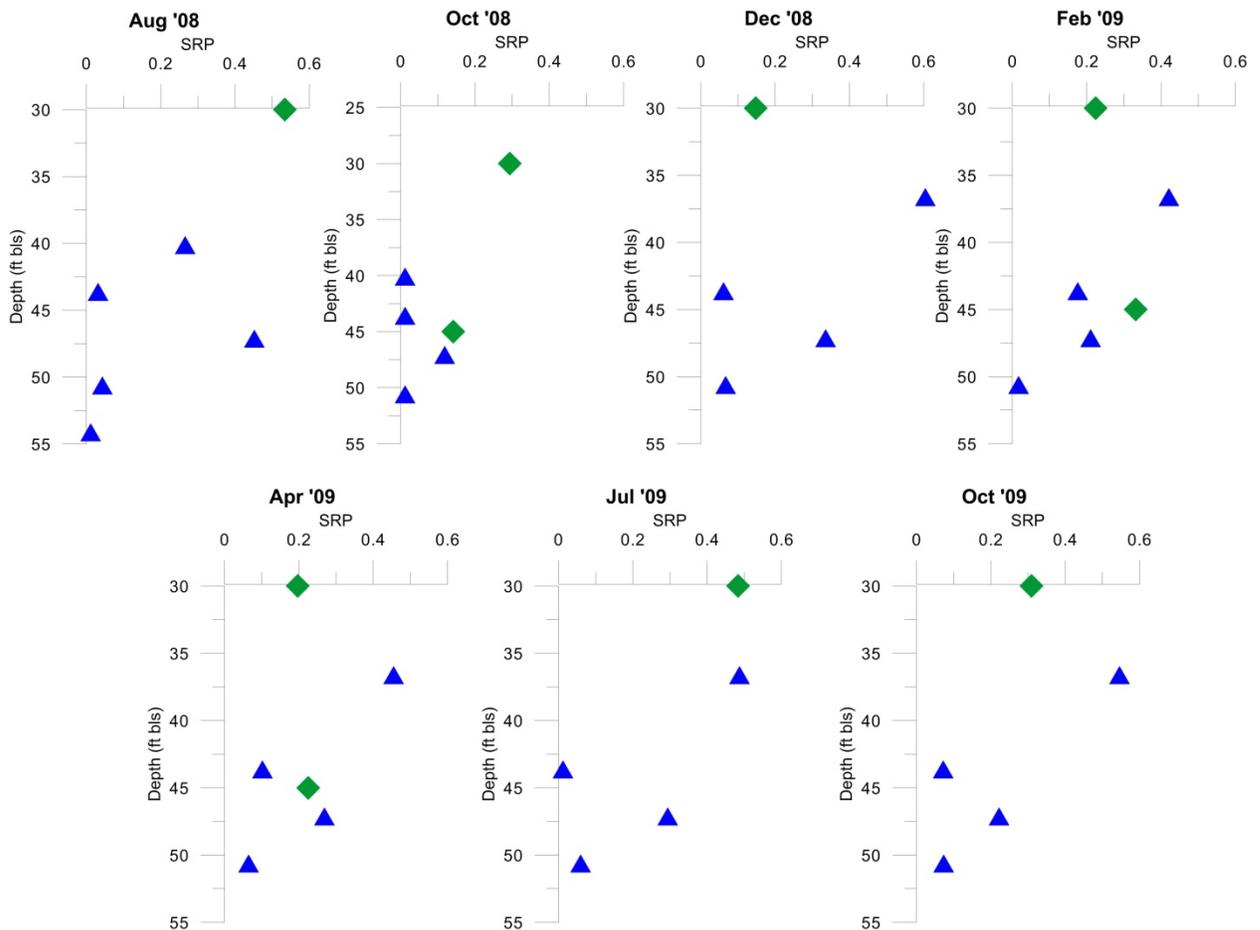


Figure 15. Orthophosphate - depth plots for standard-well Ni45-43 (diamond) and CMT1 wells Ni45-47 through 53 (triangle). Horizontal separation of wells is less than 1 m. On selected dates, the standard well was sampled at two depths with a point-source bailer.

There are no consistent correlations in NO_3^- between Ni45-35 and CMT3 ports at the same depths. During a single sampling period, there is a tendency for samples from shallow depth in the standard well to be lower than concentrations in samples from deeper CMT ports. In December 2008 and July 2009, samples from the top of the water column in Ni45-35 contained less than 10 mg/L N whereas NO_3^- concentrations in samples from deeper CMT ports are two or more times higher. This observation indicates that use of data from shallow depths in Ni45-35 would lead to an erroneous interpretation of significant reductions of NO_3^- through dispersion or denitrification.

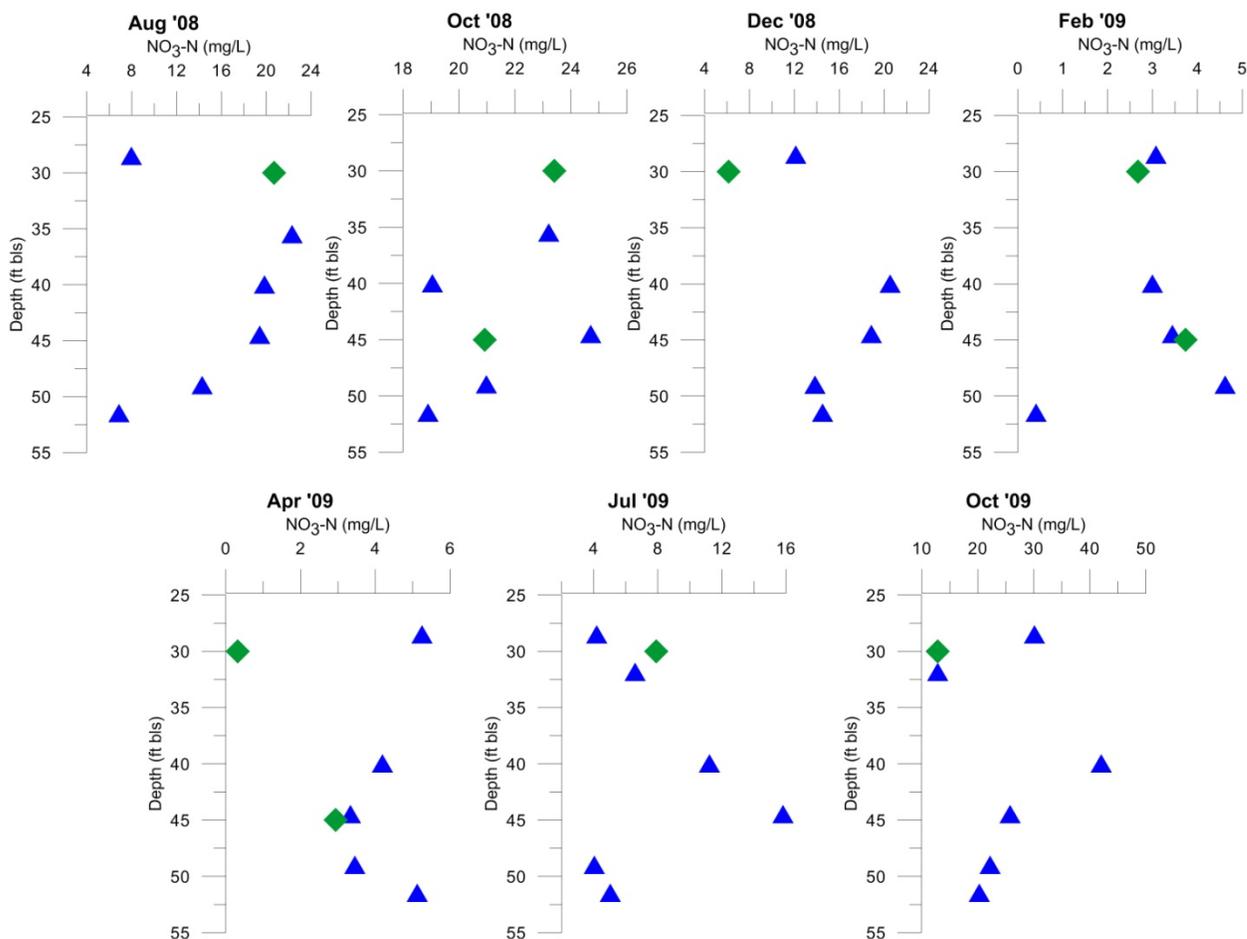


Figure 16. Nitrate-nitrogen – depth plots for standard-well Ni45-35 (diamond) and CMT3 wells Ni45-61 through 67 (triangle). Horizontal separation of wells is less than 2 m. On selected dates, the standard well was sampled at two depths with a point-source bailer.

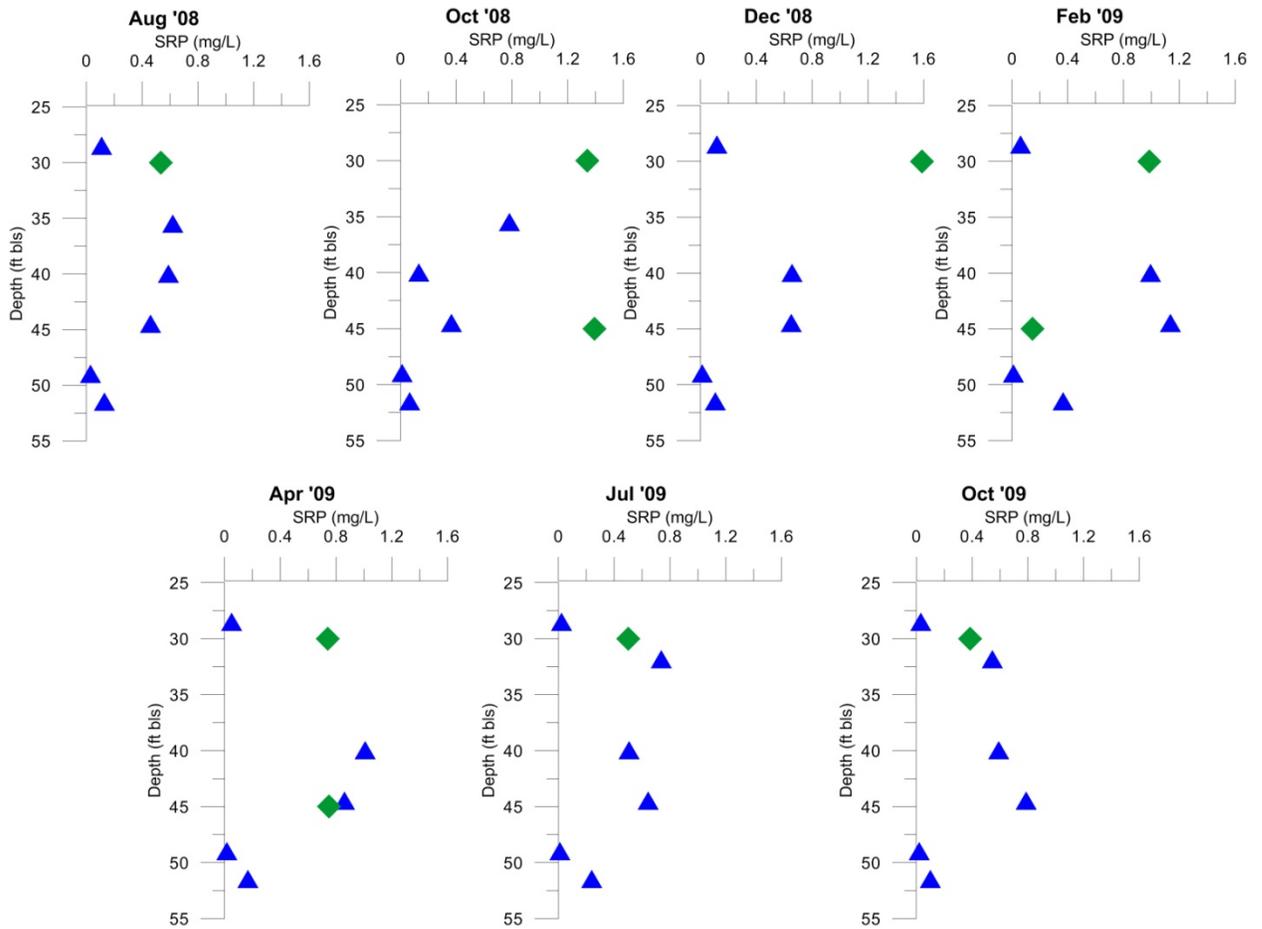


Figure 17. Orthophosphate - depth plots for standard-well Ni45-35 (diamond) and CMT3 wells Ni45-61 through 67 (triangle). Horizontal separation of wells is less than 2 m. Vertical distribution of OP changes with sampling period. On selected dates, the standard well was sampled at two depths with a point-source bailer.

Timing of P variation in Ni45-35 and CMT3 (Figure 17) are different than for NO_3^- and maximum concentrations are lower than those observed in effluent and in wells located closer to the infiltration basins (e.g., Ni45-43/CMT1). These observations are consistent with effects of sorption/desorption of P to/from the aquifer matrix. Depth pattern of P in samples from CMT ports is consistent from sample period to sample period.

There are temporally inconsistent correlations between P concentrations in Ni45-35 and CMT3 ports at the same depths. Vertical patterns of OP in CMT3 tend to remain similar between sample periods while vertical variation of OP at shallow and deep depths within

Ni45-35 change from sample period to sample period. Variability in OP between the standard well and the CMT indicate that well and sample protocols may affect results.

Well Ni45-45 and CMT2 (Ni45-54 through Ni45-60) are located about 160 ft downflow of the infiltration basins (Site 17). NO_3^- varies with depth in CMT2, with bottom-most samples consistently having the lowest concentrations (Figure 18). Smaller variations in NO_3^- are observed in shallow and deep samples from Ni45-45. Temporal variation of NO_3^- in CMT2 is not consistent from sampling period to sampling period. In contrast to the other standard well-CMT pairs, NO_3^- from similar depths in CMT and standard wells appear to be well correlated.

The highest OP concentrations are observed in mid-depth ports of CMT2 (Figure 19). The deeper sample from Ni45-45 usually has greater OP concentrations than the shallow sample. OP in CMT2 shows similar concentration versus depth profiles between sample periods. OP concentrations in Ni45-45 generally are greater than in CMT2. The magnitude of maximum concentration in Ni45-45 and CMT2 varies from sample period to sample period, with maximum concentrations in excess of 1 mg/L during four sampling periods. The presence of OP concentrations in excess of 1 mg/L indicates that loading, flow, and transport processes exceed the ability of the aquifer to sorb and store P within the 160 ft distance to the closest infiltration bed. Variability in OP between the standard well and the CMT indicate that well and sample protocols may affect results.

Standard well Ni45-42 and CMT4 (wells Ni45-78 through Ni45-84) are located in the SLS in an area where effluent does not have as significant an impact on groundwater quality as other well/CMT pairs (Site 14). Both devices are finished at much shallower depths than other well/CMT pairs. The screen in Ni45-42 is 3-ft long, not long enough to allow for collection of deep samples as was done in other standard wells.

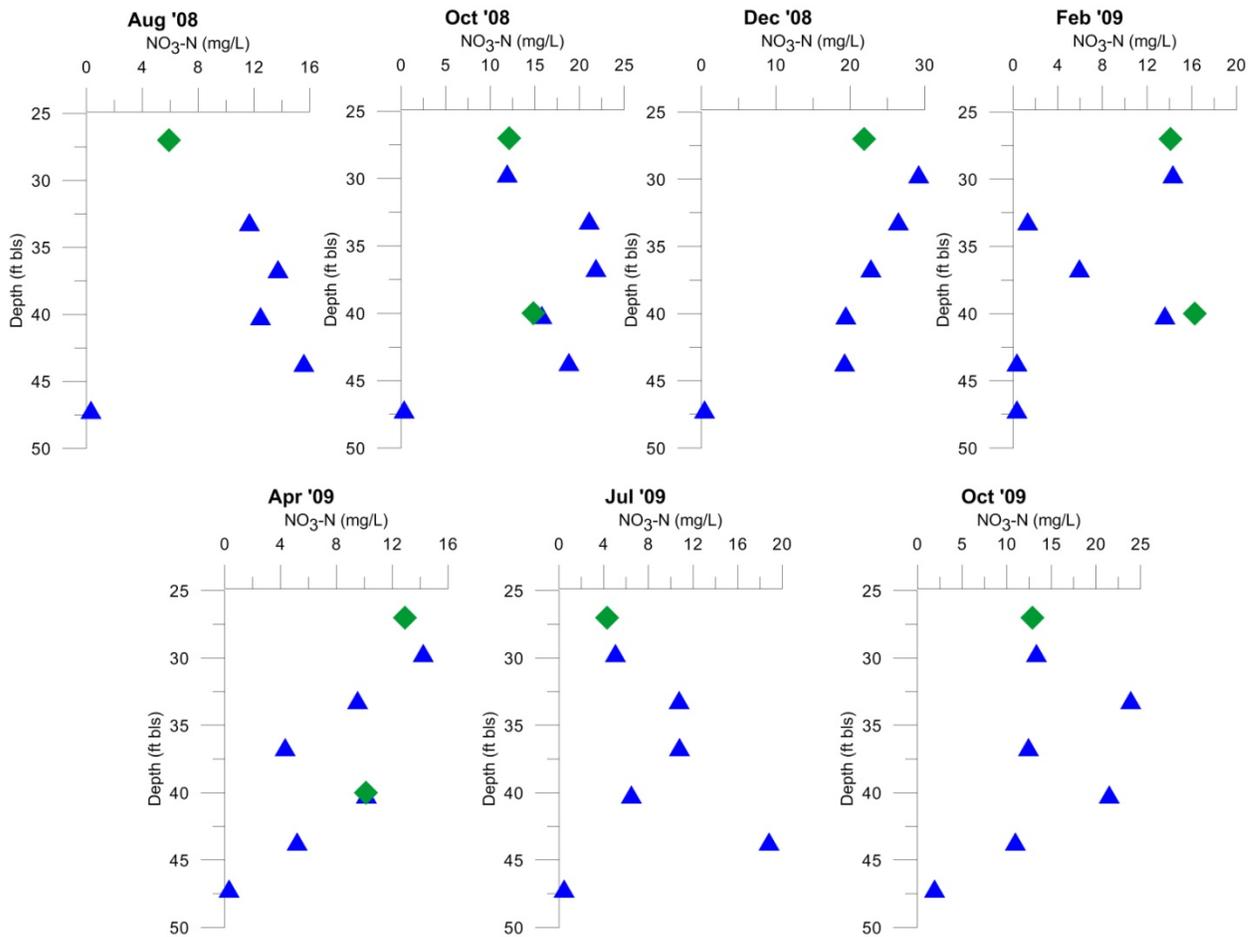


Figure 18. Nitrate-nitrogen – depth plots for standard-well Ni45-45 (diamond) and CMT2 wells Ni45-54 through 60 (triangle). Horizontal separation of wells is less than 2 m. On selected dates, the standard well was sampled at two depths with a point-source bailer.

NO₃⁻ in CMT4 tends to increase with depth and maximum concentration of NO₃⁻ greater in CMT4 than in Ni45-42 (Figure 20). This indicates that effluent flow has moved vertically downward with distance from the infiltration basins and is consistent with the interpretation that groundwater at shallow depths is likely to have recharged in areas outside of infiltration basins.

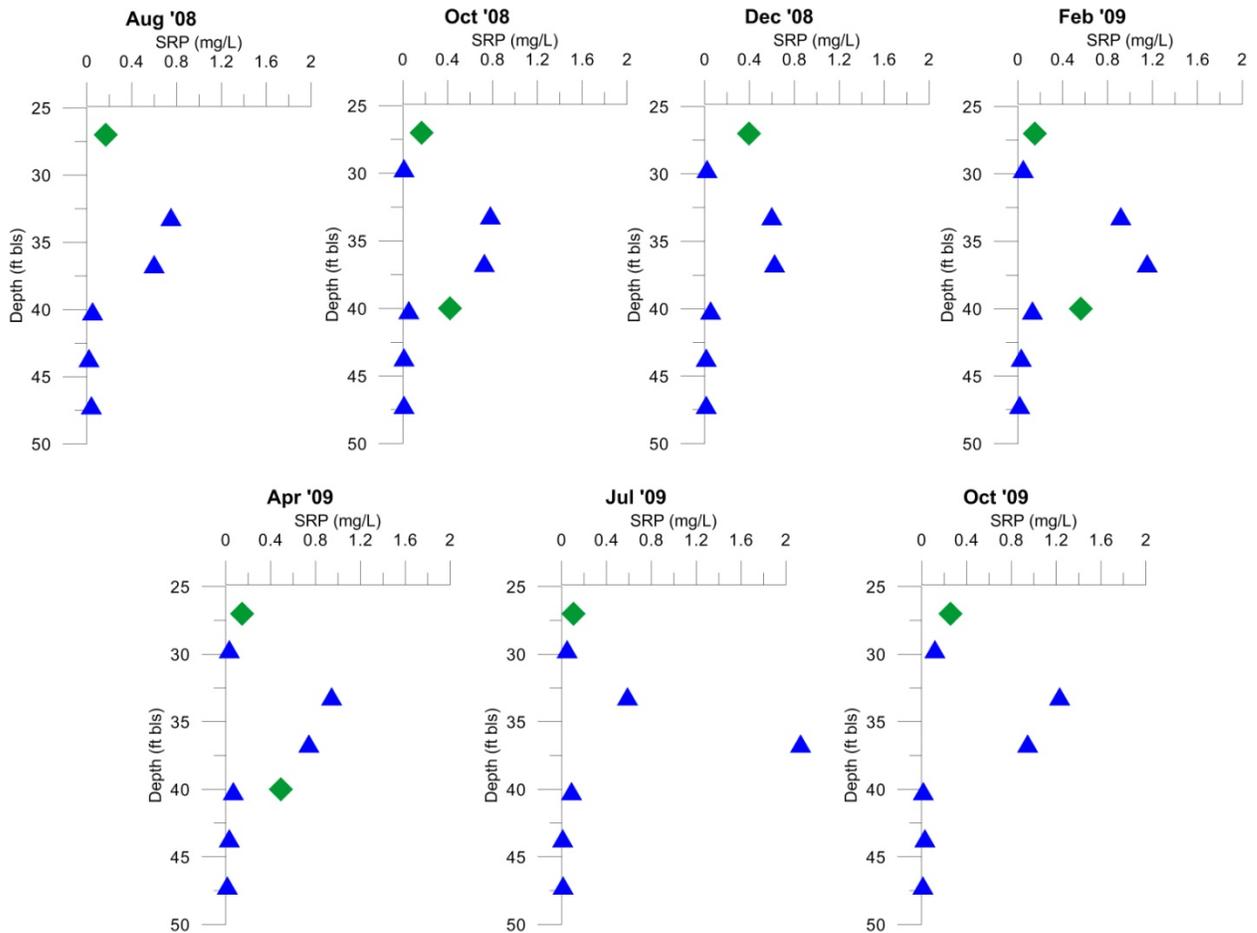


Figure 19. Orthophosphate - depth plots for standard-well Ni45-45 (diamond) and CMT2 wells Ni45-54 through 60 (triangle). Horizontal separation of wells is less than 1 m. On selected dates, the standard well was sampled at two depths with a point-source bailer.

There is significant variability in NO_3^- in samples from CMT4 during the period of February to November 2009 (Figure 20). Low NO_3^- concentrations during April and August 2009 may indicate that denitrification is occurring at this location. Denitrification would be consistent with the presence of organic-rich sediments and strong hydrogen sulfide odors noted at shallow depths in the SLS. Additional exploration at greater depth and sampling for dissolved N gas species would be the experiments that would help determine the relative importance of groundwater flow and denitrification processes in affecting the magnitude and variability of NO_3^- concentrations at this site.

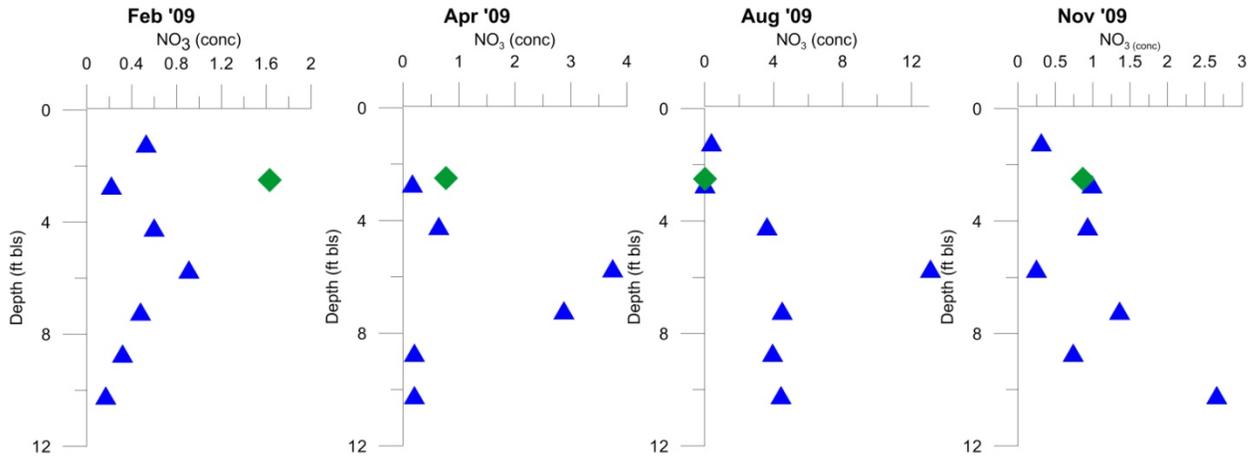


Figure 20. Nitrate-nitrogen – depth plots for standard-well Ni45-42 (diamond) and CMT4 wells Ni45-78 through 84 (triangle). Horizontal separation of wells is less than 2 m. Wells are located in the sea level swamp.

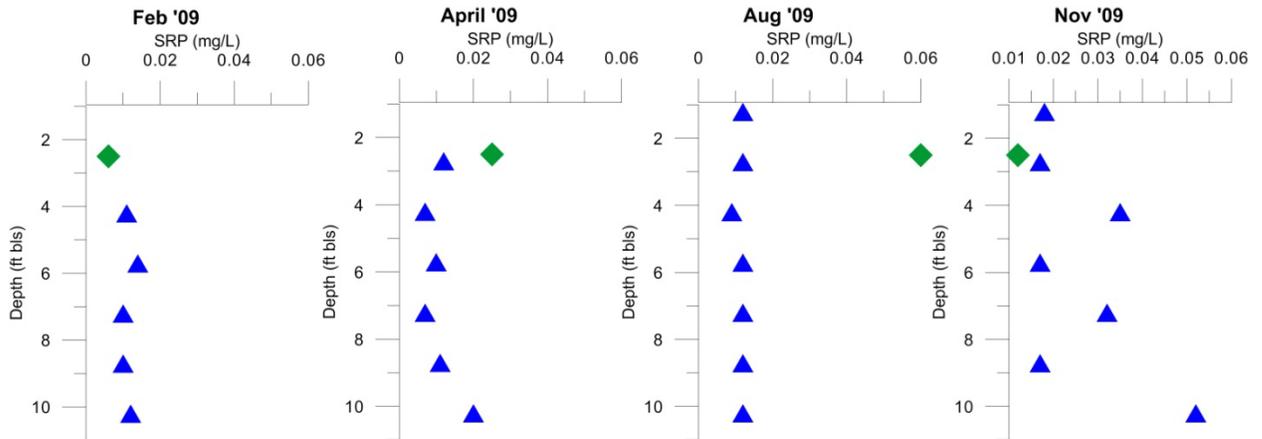


Figure 21. Orthophosphate – depth plots for standard-well Ni45-42 (diamond) and CMT4 wells Ni45-78 through 84 (triangle). Horizontal separation of wells is less than 2 m. Wells are located in the sea level swamp.

OP in CMT4 and Ni45-42 are much lower than those observed in effluent, though the observation period in CMT4 may not be long enough to detect effluent related signals. There is very little variability in P concentrations in CMT4 between sampling periods (Figure 21). There are no consistent correlations in OP between CMT4 and Ni45-42.

Denitrification and other redox sensitive processes

Denitrification, often facilitated by oxidation of organic carbon and consumption of DO, is touted as a key benefit of SAT (Crites et al., 2006, Asano et al., 2007). Denitrification

is a redox sensitive process that is favored by low DO (<0.5 mg/L), low redox potentials, and presence of sulfide minerals (McMahon and Chappelle, 2008). Strong hydrogen sulfide odors, an indicator of sulfate reduction and low redox potential, were detected in several wells (sites 9, 11, 12, 14, Figure 2) and surface water (sites 21, 22, 25-28) of the SLS. Dissolved oxygen (DO) concentrations less than 1 mg/L commonly occur in groundwater and surface water, though DO concentrations less than 0.5 mg/L, which is a threshold concentration strongly correlated with denitrification (McMahon and Chappelle, 2008) are less common. Redox potentials that are low enough to support denitrification (< 250 mV) occur in many wells on the site (Figure 23). Effluent BOD concentrations exceeded 20 mg/L in some samples, indicating an ample source of degradable organic carbon.

One indication that denitrification may be occurring is the lower concentrations of NO_3^- observed in surface water (sites 21 and 26, Figure 10) of the SLS and in some wells (sites 11 and 13, Figure 10). However, concluding that denitrification is a significant process for removing N from the surface water and groundwater of the SLS is not clear as NO_3^- concentrations vary more than an order of magnitude with time, and NO_3^- -N concentrations in excess of 10 mg/L are sometimes observed at sites that also had hydrogen sulfide, low DO (Figure 22), and low redox (Figure 23). Thus, the co-occurrence of low DO, low Eh, and NO_3^- -N concentrations greater than 10 mg/L could indicate complex mixing of anaerobic, reduced waters with more oxidized waters, with mixing caused by small scale heterogeneities in hydraulic conductivity and aquifer matrix composition rather than denitrification (Jurgens et al., 2009).

It is also possible that both well and surface-water sampling points exhibiting lower NO_3^- concentrations are not in locations where groundwater with high NO_3^- concentrations is flowing. At sites 11, 13, and 21, at the toe of the dune (Figure 10), NO_3^- concentrations in most samples are less than 1 mg/L, and low DO, low Eh, and hydrogen sulfide were noted. However, SC at site 11 is less than that observed in samples from other wells in the preferential flow zone indicating that this well is not impacted by effluent. Given the greater NO_3^- concentrations at depth in the sampling ports in CMT4 (Figure 20, site 14 in

the SLS), it is likely that the well at site 11 is not deep enough to intersect effluent impacted groundwater.

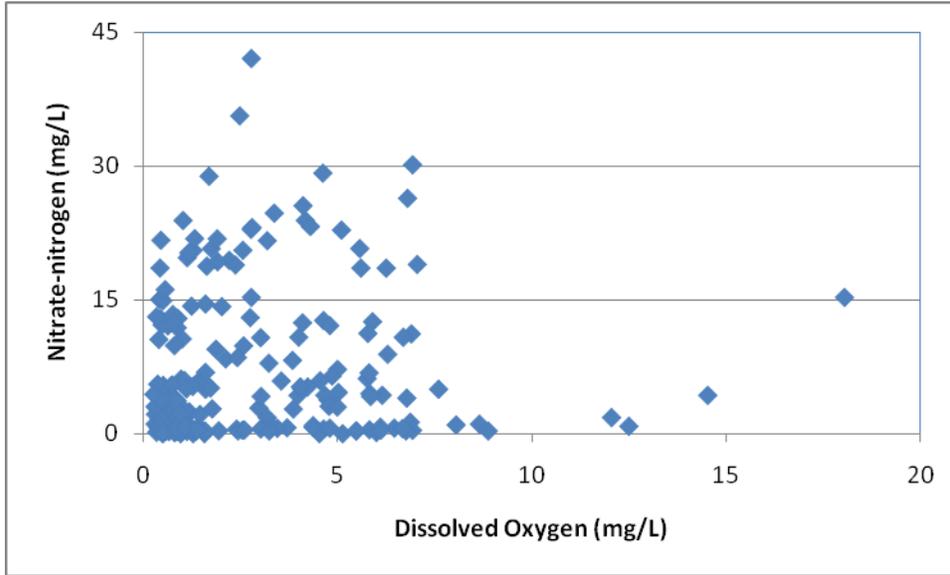


Figure 22. Comparison of dissolved oxygen and nitrate for surface water and groundwater. Note that nitrate-nitrogen concentrations in excess of 10 mg/L occur at dissolved oxygen concentrations less than 1 mg/L.

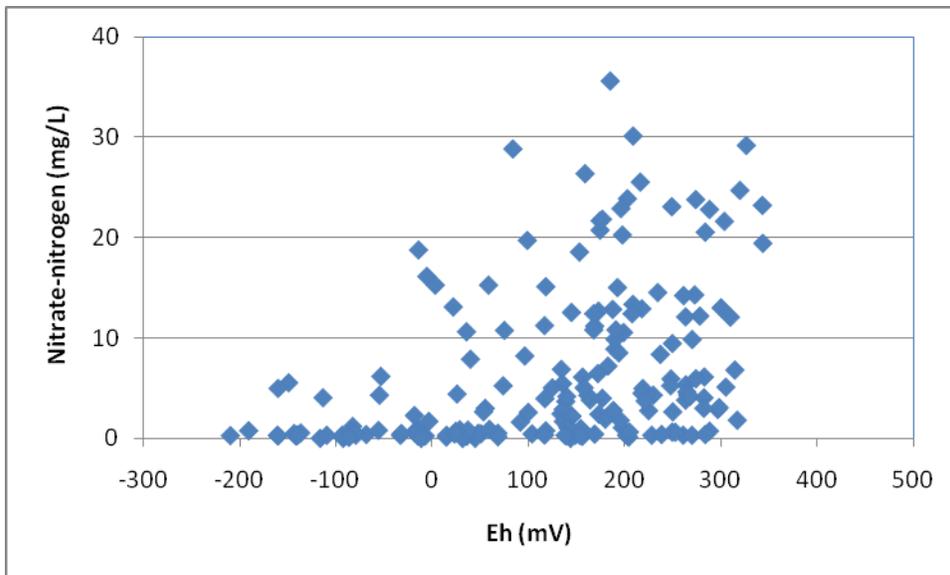


Figure 23. Comparison of redox potential and nitrate in surface water and groundwater. Redox potential (Eh) is reported in millivolts (mV). Note that nitrate-nitrogen concentrations greater than 10 mg/L occur at Eh values less than 250 mV, that being the approximate value where denitrification should occur.

Specific conductance at site 13 is similar to that observed in other wells in the preferential flow zone suggesting that groundwater at this site may be affected by effluent disposal. The higher SC however, may be due to the higher Cl concentrations in this well.

Data collected during this study are not adequate to determine if denitrification is occurring in the aquifer at this site. If denitrification is occurring, then spatially and temporally variable concentrations of NO_3^- in the aquifer and surface water indicates that the rate of denitrification is spatially and temporally variable. An adequate experiment for estimating denitrification rates should include additional rounds of sampling and testing for concentrations of nitrogen gas and multi-month tracer tests using Br or other conservative tracers added to the effluent.

Significant concentrations of dissolved iron, arsenic, and OP in groundwater are also indicators of redox sensitive biogeochemical reactions (Herbel and Fendorf, 2006; McMahon and Chappelle, 2008; Vadas et al., 2007; Bell et al., 2009). Similar relationships have been observed in soils for P and iron (Vadas and Sims, 1998, 1999; Sallade and Sims, 1997) and arsenic (Sparks et al., 2007) In some cases, arsenic sorbed to iron oxide grain coatings can be released to solution when biogeochemical processes cause iron and/or manganese to be reduced (Herbel and Fendorf, 2006; Haque et al., 2008).

Data collected during this study clearly indicate that redox related processes are affecting geochemical conditions; however, the relationships are complex and indicate that redox conditions are highly spatially variable. For example, comparison of iron and arsenic with DO and redox potential (Figures 24-28) show no clear correlations that would be expected if the entire aquifer were strongly reduced. The lack of clear correlations indicates that multiple redox processes and/or mixing of different water bodies in a heterogeneous geologic setting are controlling the distribution of DO, OP, iron, and arsenic.

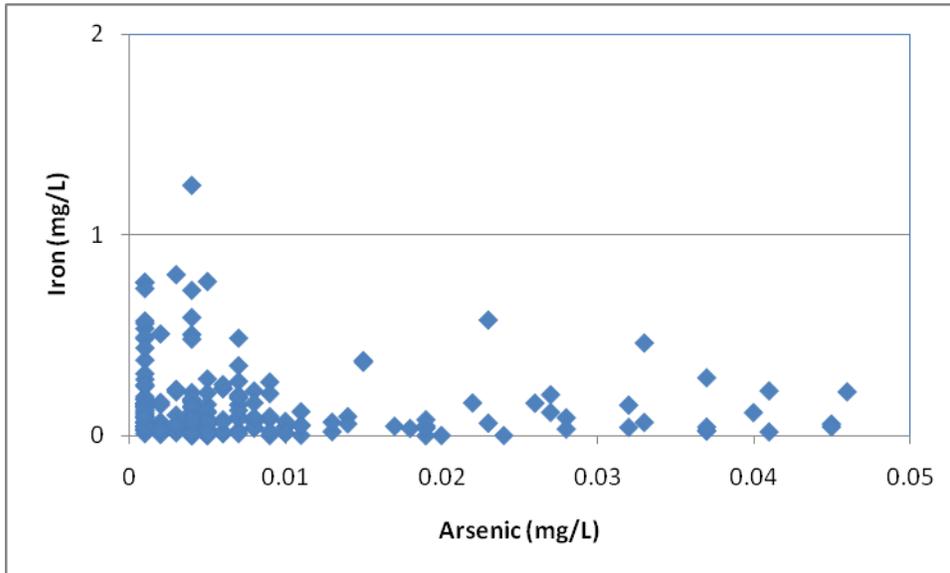


Figure 24. Comparison of dissolved iron and dissolved arsenic in groundwater and surface water.

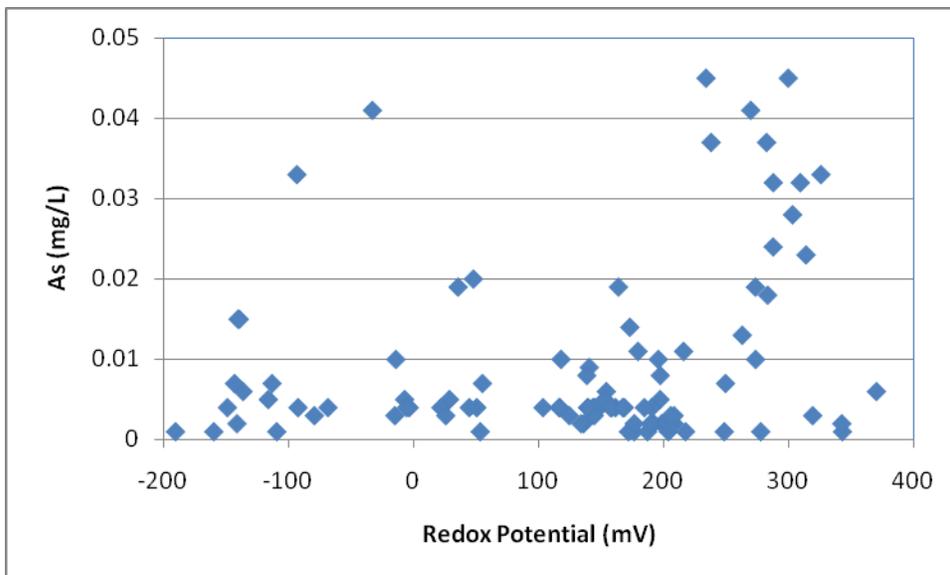


Figure 25. Comparison of redox potential and dissolved arsenic in groundwater and surface water. mV=millivolts, mg/L=milligrams per liter.

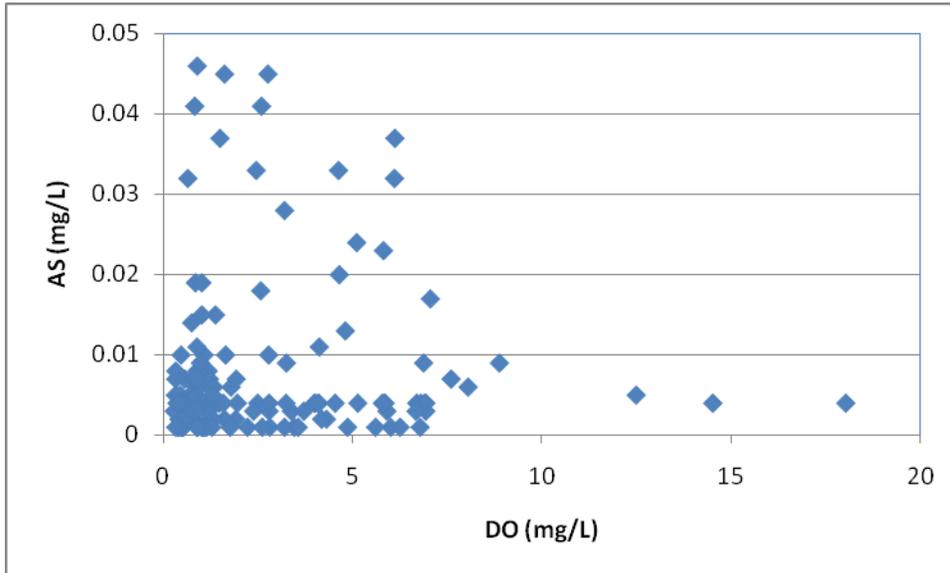


Figure 26. Comparison of dissolved oxygen and dissolved arsenic in groundwater and surface water.

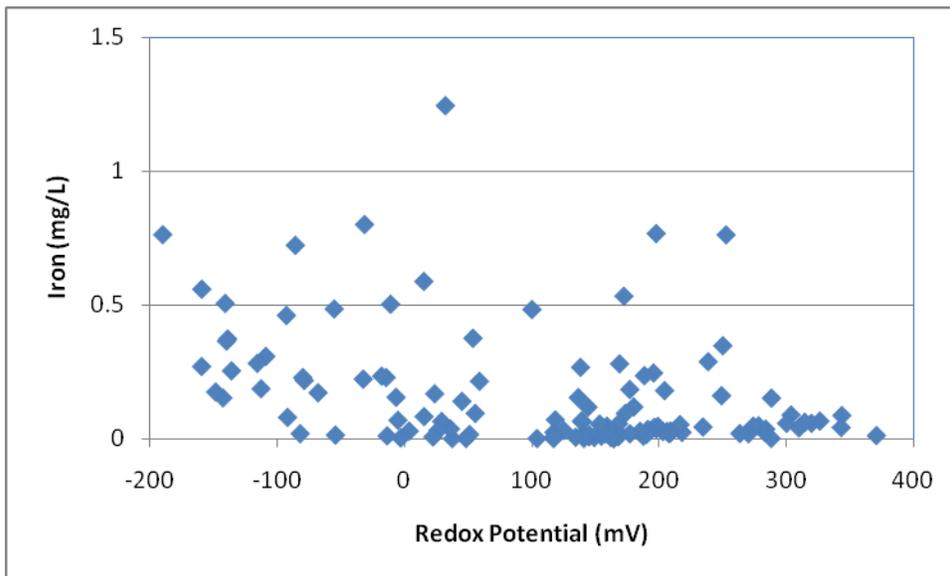


Figure 27. Plot of redox potential and dissolved iron in groundwater and surface water.

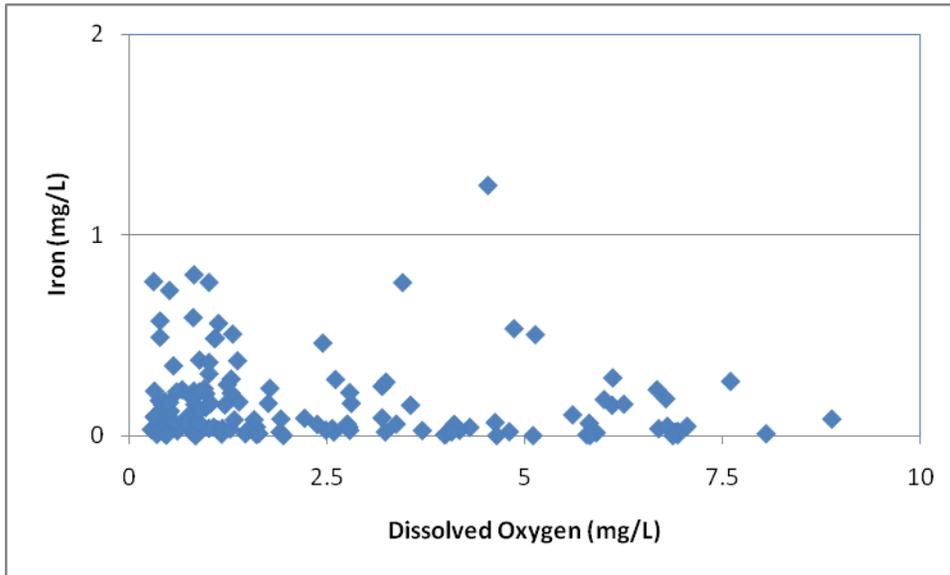


Figure 28. Comparison of dissolved oxygen and dissolved iron in groundwater and surface water.

Significant concentrations of dissolved arsenic are observed in many samples (Figure 29). Laboratory experiments have found that OP can outcompete arsenic for sorption sites on iron oxide coatings leading to desorption of arsenic or selective sorption of OP (Sracek et al., 2004), though it is not clear how results from a laboratory experiment apply to a complex groundwater environment. Data from this study indicate no clear relationship between arsenic and OP (Figure 30). It is also not clear if ionic strength (via SC proxy, Figure 31) facilitates desorption of arsenic and increasing concentration. The lack of clear correlations indicates that multiple processes and/or mixing of different water bodies in a heterogeneous geologic setting are controlling the distribution of arsenic. These findings indicate the need for further study.

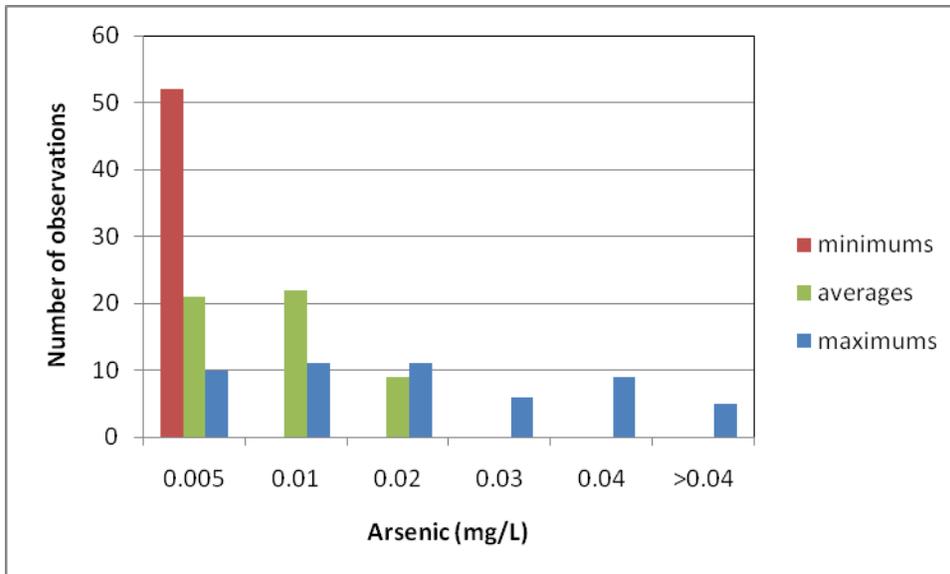


Figure 29. Frequency distribution of arsenic concentrations aggregated by sample location. Average concentrations were computed assuming that concentrations in samples reported less than detect are equal to the detection limit. The total number of sample sites is 52.

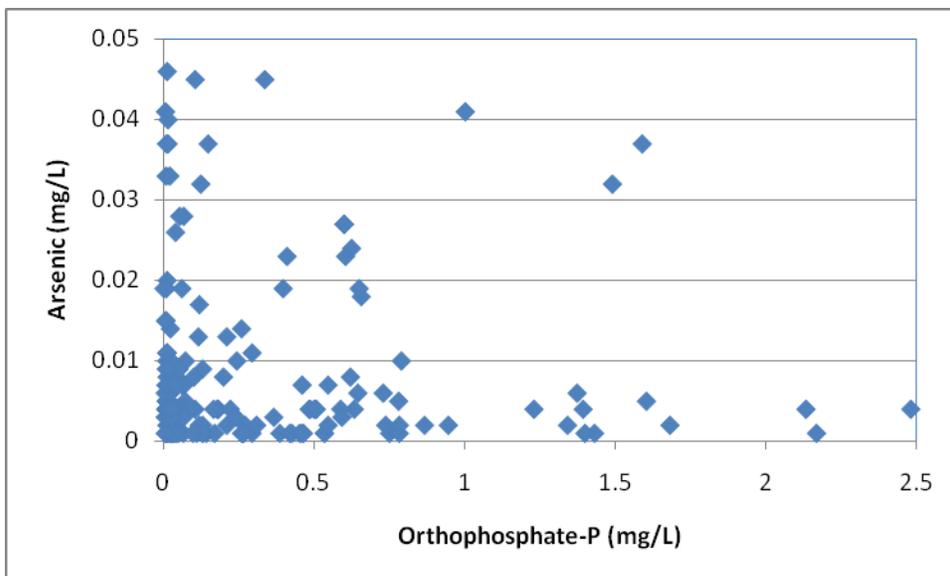


Figure 30. Comparison of orthophosphate and dissolved arsenic in groundwater and surface water.

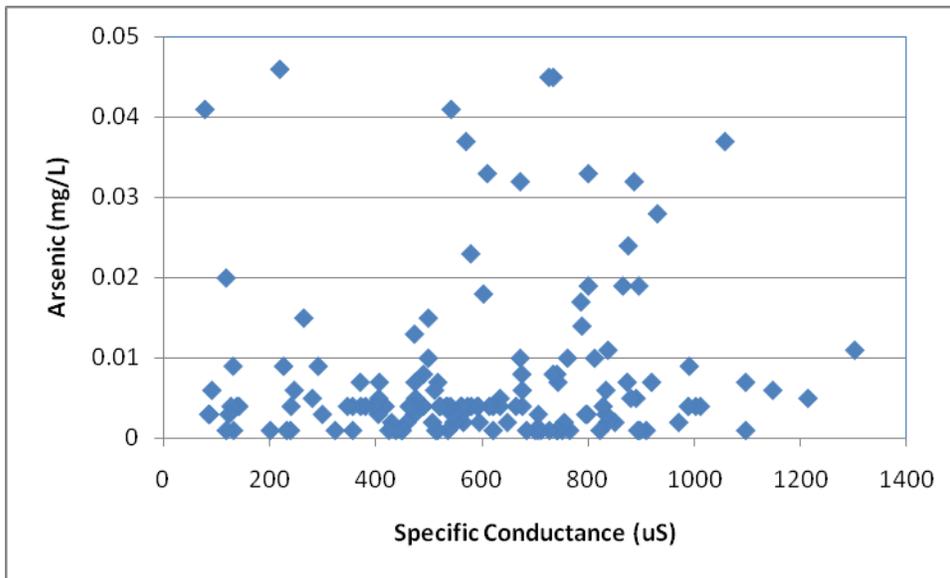


Figure 31. Comparison of specific conductance and dissolved arsenic in groundwater and surface water.

Influence of monitoring methods and effluent disposal on observed groundwater quality

Because of high hydraulic loading rates and water-table mounding associated with RIBS, groundwater flow paths near RIBS have an accentuated three-dimensional aspect compared to areas receiving natural rates of recharge. As a result, there is a significant chance that bailer or low-flow pump sampling of long-screen wells will indicate a biased representation of the presence and concentration of contaminants in the aquifer (i.e., false negative). In our study, results indicate that sampling at too shallow a depth increases the likelihood of not detecting contaminants at depth. This likelihood increases with increasing distance from the infiltration basins. Our study also indicates the likelihood that contaminants may not be detected by long-screen wells sampled by low-flow sampling methods when samples are collected from portions of the aquifer below that affected by effluent disposal.

Our study also found that there is significant horizontal and vertical variability of contaminant concentrations within the portion of the aquifer most impacted by effluent disposal (e.g., preferential flow zone, Figure 32). Despite the relatively small spatial extent of the disposal area in our study area, identification of the preferential flow zone

and characterization of the vertical and temporal variability in the concentrations of contaminants required a multi-phase subsurface investigation program that included analysis of data from samples collected at bi-monthly intervals from dozens of monitoring points and high frequency temperature monitoring in several wells.

Following practices used in many studies of groundwater containing hazardous materials, we used a chemically inert tracer (e.g. temperature) to independently confirm the location of the preferential flow zone where the bulk of effluent is flowing in the aquifer. Further analysis of the temperature signal is not possible at this time because physical interactions between effluent and the aquifer matrix dissipate heat, and we do not have independent measures of the thermal conductivity of the aquifer matrix. Temperature cannot be used as a tracer of contamination at all sites because not all effluents have a temperature signature. In many cases, bromide is commonly used as the chemically inert tracer (citations in “Contaminant transport” in Results of Literature Search).

Dissolved N or P species are biogeochemically reactive species and as such, should not be used as tracers. By extension, the lack of N and P species in groundwater samples is not confirmation of contaminant attenuation by SAT. Rather, lack of N and P species in groundwater samples can equally reflect improper spatial location of the monitoring sites, inadequate temporal frequency of sampling, and/or effects of well and sampling protocols. For the latter, the choice of well type and sampling method used for standard wells may introduce more variability to observed P concentrations than the method used for CMT ports. It is possible that the slightly higher flow rate during purging of the standard well and greater agitation of the water column caused by insertion and removal of the bailer from the standard well could mobilize P sorbed on the aquifer matrix. In this case, it would be important to know if the P sorbed to the aquifer matrix is naturally occurring or due to effluent disposal. It is also possible that P mobility can be reduced by temporally variable geochemical conditions in the aquifer due to oxygenation of the water column in the well between sampling periods or during insertion and removal of pumps and bailers. It cannot be determined if purging greater volumes of water from wells prior to sample collection would reduce this effect.

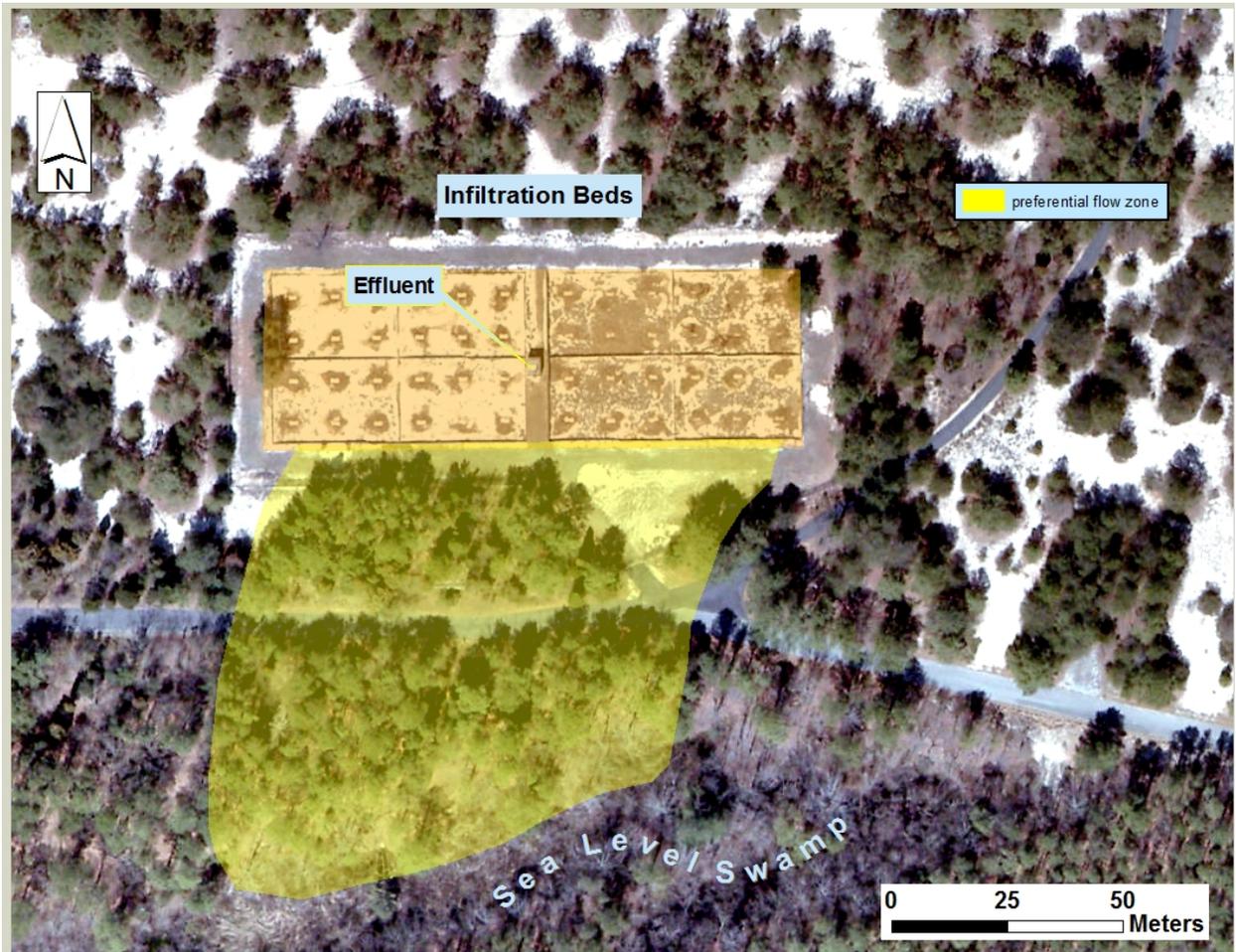


Figure 32. Conceptual model of preferential flow zone. Concentrations of N and P and temperatures are generally greater in groundwater in this zone than outside of this zone.

This study confirms that contaminant transport from RIBS/SAT is subject to the same physical and chemical processes that have been observed in many field studies of groundwater contamination by hazardous and toxic materials; that is, heterogeneous hydraulic and geochemical aquifer properties, and time-varying contaminant releases cause complex spatial and temporal patterns of contaminants in the aquifer. Because of these complexities, detailed field studies are needed to 1) determine where effluent is moving in the aquifer and how water quality changes with time and distance from the disposal site, 2) determine if serious contamination is occurring, and 3) to determine how quickly contaminants are flowing away from a RIBS site. Simple monitoring systems consisting of a few long-screen wells that are sampled four times per year at low-flow

rates, and with well locations and screen settings determined prior to collection and analysis of site-specific hydraulic, geochemical, and tracer test data greatly increase the risk of not detecting contamination problems before they impact downflow wells and streams.

At this site, drill-rig access and resource limitations did not allow for installation of monitoring wells around the west-southwestern, northwestern, and northeastern sections of the infiltration basins. As a result, we cannot be absolutely certain that contaminants are not flowing in those directions. Particle trackline modeling in Andres et al. (2010), however, indicates that flow paths oriented in those directions are relatively minor and should not transport significant quantities of contaminants in those directions. This topic will be addressed in an upcoming report on results of detailed three-dimensional groundwater flow simulations of this facility.

CONCLUSIONS

Significant research is being conducted and published in peer reviewed technical literature on specific topics that directly apply to RIBS, new and novel wastewater treatment systems, infiltration, transport of contaminants in groundwater, groundwater monitoring, natural attenuation and remediation, and affects of land-based wastewater disposal on groundwater. Many citations of RIBS-specific research appear in non-reviewed literature such as conference proceedings, newsletters, and trade journals.

Most of the background peer-reviewed research pertaining to RIBS cited by USEPA guidance documents and several textbooks was conducted in the 1970s using methods and monitoring systems that would be considered inadequate by today's standards to characterize the physical and chemical conditions in the groundwater beneath a RIBS facility. Review of these documents indicates that nitrogen and phosphorus removal statistics are highly suspect.

The effects of RIBS on groundwater are highly complex. Spatially heterogeneous aquifer properties, temporal variability of quality and quantity of effluent, and changes in

disposal location associated with operation of RIBS, cause spatially and temporally complex variability of physical and chemical groundwater conditions.

A monitoring system includes wells, surface water sampling stations, automated instrumentation, sample collection, and analyses of sediments and water. Simple monitoring systems consisting of a few wells sampled at prescribed quarterly intervals and tested for a limited suite of physical and chemical parameters are highly likely to provide data biased towards underestimating the impact of effluent disposal on groundwater quality.

A detailed comparative study of standard and multi-port groundwater monitoring systems indicates that bailer or low-flow pump sampling of standard monitoring wells is also highly likely to provide data biased towards underestimating the impact of effluent disposal on groundwater quality.

A well designed monitoring system should be based on experimentally determined site-specific evidence collected under conditions that duplicate flow rates to be expected during full-scale operation of the RIBS. This type of test should include use of conservative tracers to determine if the monitoring wells are placed in locations to intercept flow from the infiltration beds. The monitoring system for any facility should consider site-specific biogeochemical interactions of effluent with geologic materials, direction and rate of ground water flow, sources and behaviors of complex mixtures of contaminants, expected variability in effluent flow and quality, and impacts to sensitive receptors. Monitoring systems designed on the basis of assumptions of average effluent flow and quality characteristics, average operation of the infiltration basins, average rate of flow for a single conservative contaminant, and no potential geochemical interactions with the aquifer matrix will almost certainly not be adequate to determine the impact of the operational RIBS.

Significant concentrations of nitrate, ammonium, and organic N occur in groundwater from point of effluent entry at the water table and in surface water in the adjacent

discharge area. Data are not sufficient to determine how much denitrification is occurring at Cape Henlopen State Park. If denitrification is occurring, then the rate of denitrification is insufficient to remediate RIBS effluent at the site — this despite a 25-ft thick vadose zone, an effluent with ample organic carbon to facilitate anaerobic conditions that permit abiotic denitrification and feed microorganism-driven denitrification processes, and hypoxic to anoxic groundwater. Given recent efforts to increase beneficial reuse of wastewater and infiltration of stormwater, and new plans for large on-site wastewater disposal facilities, additional field research on denitrification at Delaware on-site wastewater disposal facilities is warranted.

Significant concentrations of P occur in groundwater from point of effluent entry at the water table to distances greater than 150 ft from the infiltration beds. It appears that high hydraulic, P, and organic loading rates associated with the operation of RIBS overwhelms natural P attenuation (e.g., sorption and precipitation) processes. It is prudent to assume that knowledge and experience gained from decades of study P loss from agricultural systems in Delaware should have direct application to this issue, just as it is important to consider that the Delaware Nutrient Management Commission continues to spend significant financial resources to mitigate P loss from agricultural systems. Given recent efforts to increase beneficial reuse of wastewater and infiltration of storm water, and new plans for large on-site wastewater disposal facilities, additional field research on how to adapt agronomic testing and planning practices for these facilities is warranted.

Because of problems with trihalomethane (THM) generation and aquifer contamination at other RIBS facilities caused by chlorine disinfection, strong consideration should be given to not permitting this practice. If chlorine-based disinfection is necessary, additional research is warranted to document the fate and transport of THMs and other disinfection byproducts associated with discharge of chlorinated effluent to RIBS. Similar issues associated with the myriad of bacteria and viruses, and pharmaceutical, personal care, and household products that enter wastewater treatment and disposal systems also indicate the need for additional field research.

Interactions of effluent with the aquifer and native groundwater appear to result in geochemical changes that cause the release of naturally occurring arsenic, and other redox-sensitive compounds and elements from the aquifer matrix to the groundwater. A separate, parallel study is addressing the occurrence of arsenic and other redox-sensitive elements in a wider range of Coastal Plain sediments, and those results and additional recommendations will be made available when the work is complete.

REFERENCES CITED

- AFCEE, 2008, Ashumet Pond geochemical barrier project: Air Force Center for Engineering and the Environment.
- Andres, A.S., 1991, Results of the Coastal Sussex County, Delaware Ground-Water Quality Survey: Delaware Geological Survey Report of Investigation No. 49, 28 p.
- Andres, A.S., 1995, Nitrate loss via groundwater flow, coastal Sussex County, Delaware, p. 69-76. *In* K. Steele (ed.), *Animal Waste and the Land-Water Interface*. CRC Press, Boca Raton, Florida
- Andres, A. S., and Klingbeil, A. D., 2006, Thickness and transmissivity of the unconfined aquifer of eastern Sussex County, Delaware: Delaware Geological Survey Report of Investigations No. 70, 19 p.
- Andres, A. S., Walther, E. F., Turkmen, M., and He, C., 2010, Hydrogeology of a rapid infiltration basin system at Cape Henlopen State Park, Delaware: Delaware Geological Survey unpublished contract report to DNREC, 41 p.
- Asano, T., Burton, F.L., Leverenz, H.L., Tsuchihashi, R., and Tchobanoglous, G., 2007, *Water Reuse; Issues, Technologies and Applications*: New York, NY, McGraw-Hill, 1570 p.
- Ator, S. W., 2008, *Natural and Human Influences on Water Quality in a Shallow Regional Unconsolidated Aquifer, Northern Atlantic Coastal Plain*, U.S. Geological Survey Scientific Investigations Report 2008-5190, 32 p.
- Aulenbach, D. B., 1979, Long-term recharge of trickling filter effluent into sand: USEPA, EPA600/2-79-068, 146 p.
- Aulenbach, D.B., and Clesceri, N.L., 1980, Monitoring for land application of wastewater: *Water, Air, & Soil Pollution*, v. 14, p. 81-94.
- Bachman, L.J., 1984, Nitrate in the Columbia Aquifer, central Delmarva Peninsula, Maryland: U. S. Geological Survey, Water Resources Investigation Report 84-4322, 51 p.
- Bales, R. C., Li, S., Yeh, T. C., Lenczewski, M. E., Gerba, C. P., 1997, Bacteriophage and microsphere transport in saturated porous media: Forced-gradient experiments at Borden, Ontario: *Water Resources Research*, v. 33, p. 639-648.
- Banks, W. J., and Klohe, C. A., 2001, Occurrence and distribution of enteric viruses in shallow ground water and factors affecting well vulnerability to microbiological contamination in Worcester and Wicomico Counties, Maryland: U.S. Geological Survey Water-Resources Investigation Report 01-4147, 23 p.
- Baker, M.J., Blowes, D.W., Ptacek, C.J., 1998, Laboratory development of permeable reactive mixtures for the removal of phosphorus from onsite wastewater disposal systems. *Environ. Science and Technology* 32, 2308–2316.
- Bell, K. Y., Wiseman, L., and Turner, L. A., 2009, Designing pretreatment to control arsenic leaching in ASR facilities: *Journal American Water Works Association*, v. 101, p. 74-84.
- Benham-Blair, and Affiliates, Inc., and Engineering Enterprises, Inc., 1979, Long-term effects of land application of domestic wastewater, Milton, Wisconsin, rapid infiltration site: USEPA, EPA600/2-78-145, 128 p.

Bolton, D. W., 2000, Occurrence and distribution of radium, gross alpha-particle activity, and gross beta-particle activity in ground water in the Magothy Formation and Potomac Group aquifers, upper Chesapeake Bay area, Maryland: Maryland Geological Survey Report of Investigations No. 70, 97 p.

Brisbin, T. D., Ahn, S. H., Foster, R. I., Labunski, S. A., and Oliva, J. A., 1984, Priority pollutants in the Cedar Creek reclamation-recharge facilities: USEPA, EPA-600/S2-84-061, 5 p.

[Brooks, J.L.](#), [Rock, C.A.](#), and [Struchtemeyer, R.A.](#), 1984, Use of peat for on-site wastewater treatment: II. Field studies: *Journal of Environmental Quality*, v. 13, p. 524-530,

Burde, M., Rolf, F, and Grabowski, F., 2001, Innovative low cost procedure for nutrient removal as an integrated element of a decentralised water management concept for rural areas: *Water Science and Technology*, v. 44, p. 105-112.

Center for the Inland Bays, 1995, A comprehensive conservation and management plan for Delaware's Inland Bays, Chapters 1-5: Rehoboth Beach, DE, Center for the Inland Bays, 132 p.

Commission on Geosciences, Environment, and Resources, 1994, Alternatives for ground water cleanup: National Academies Press, 316 p.

Crites, R.W., Middlebrooks, E.J., and Reed, S.C., 2006, *Natural Wastewater Treatment Systems*: New York, NY, Taylor & Francis Group, 552 p.

Crites, R. W., Reed, S.C., and Bastian, R.K., 2000, *Land Treatment Systems for Municipal and Industrial Wastes*: New York, McGraw-Hill, 433 p.

Crites, R.W., and Tchobanoglous, G., 1998, *Small and Decentralized Wastewater Management Systems*: New York, McGraw-Hill, 1084 p.

Day-Lewis, F. D., Lane, J. W., and Gorelick, S. M., 2004, Combined interpretation of radar, hydraulic, and tracer data from a fractured-rock aquifer near Mirror Lake, New Hampshire, USA: *Hydrogeology Journal*, v. 14, p. 1-14.

Debrewer, L. M., Ator, S. W., and Denver, J. M., 2005, Factors Affecting Spatial and Temporal Variability in Nutrient and Pesticide Concentrations in the Surficial Aquifer on the Delmarva Peninsula, U.S. Geological Survey Scientific Investigations Report 2005-5257, 56 p.

Denver, J. M., 1989, Effects of agricultural practices and septic-system effluent on the quality of water in the unconfined aquifer of eastern Sussex County, Delaware: Delaware Geological Survey Report of Investigation No. 45, 66 p.

Denver, J.M., Ator, S.W., Debrewer, L.M., Ferrari, M.J., Barbaro, J.R., Hancock, T.C., Brayton, M.J., and Nardi, M.R., 2004, *Water Quality in the Delmarva Peninsula, Delaware, Maryland, and Virginia, 1999–2001*: U.S. Geological Survey Circular 1228, 27 p.

Einarson, M. D., and Cherry, J. A., 2002, A new multi-level ground water monitoring system using multichannel tubing: *Ground Water Monitoring and Remediation*, v. 22, p. 52-65.

Enfield, C. G., Phan, T., Walters, D. M., and Ellis, R., 1981a, Kinetic model for phosphate transformation in calcareous soils: I. Kinetics of Transformation: *Soil Science Society of America Journal*, v. 45, p. 1059-1064.

Enfield, C. G., Phan, T., Walters, D. M., 1981b, Kinetic model for phosphate transformation in calcareous soils: II. Laboratory and field transport: *Soil Science Society of America Journal*, v. 45, p. 1064-1071.

Ferrari, M. J., 2001, Occurrence and Distribution of Selected Contaminants in Public Drinking-Water Supplies in the Surficial Aquifer in Delaware, U.S. Geological Survey Open-File Report 01—327, 17p.

Freeze, R. A., and Cherry, J. A., 1979, Groundwater: Englewood Cliffs, NJ, Prentice-Hall, 604 p.

Fox, P., and Gable, J., 2003, Sustainable nitrogen removal by anaerobic ammonia oxidation during soil aquifer treatment: Water Environment Federation, Water Environment Federation, WEFTEC 2003, p. 77-87.

[Green, M.](#), [Tarre, S.](#), [Schnizer, M.](#), [Bogdan, B.](#), [Armon, R.](#), [Shelef, G.](#), 1994, Groundwater denitrification using an upflow sludge blanket reactor: Water Research, v. 28, p. 631-637.

Greenberg, A. E., Clesceri, L. S., Trussell, R. R., eds., 1998. Standard methods for the examination of water and wastewater. 20th ed., American Public Health Association, American Water Works Association and Water Pollution Control Federation, Washington, D.C., pp.4-75-4-81.

Guitierrez-Magness, A. L., and Raffensperger, J. P., 2003, *Development, Calibration, and Analysis of a Hydrologic and Water-Quality Model of the Delaware Inland Bays Watershed*, U.S. Geological Survey Water Resources Investigation Report 03-4124, 46 p.

Haque, S., Ji, J., and Johannesson, K. H., 2008, Evaluating mobilization and transport of arsenic in sediments and groundwaters of Aquia aquifer, Maryland, USA: Journal of Contaminant Hydrology, v. 99, p. 68-84.

Hem, J.D., 1992, Study and interpretation of chemical characteristics of natural water (3d ed.): U.S. Geological Survey Water-Supply Paper 2254, 263 p.

Herbel, M., and Fendorf, S., 2006, Biogeochemical processes controlling the speciation and transport of arsenic within iron coated sands: Chemical Geology, v. 228, p. 16-32.

Hu, C., Zhang, T., Huang, Y., Dahob, M., and Surampalli, R., 2005, Effects of long-term wastewater application on chemical properties and Phosphorus adsorption capacity in soils of a wastewater land treatment system: Environmental Science and Technology v. 39, p. 7240-7245.

Hubbard, S. S., Chen, J., Peterson, J., Majer, E. L., Williams, K. H., Swift, D. J., Mailloux, B., and Rubin, Y., 2001, Hydrogeological characterization of the South Oyster Bacterial transport site using geophysical data: Water Resources Research, v. 37, p. 2431-2456.

Johnson A. I., and Pyne, D. G., eds., 1994, Artificial recharge of ground water, II: New York, ASCE, 913 p.

Jurgens, B. C., McMahon, P. B., Chappelle, F. H., and Eberts, S. M., 2009, An Excel Workbook for Identifying Redox Processes in Ground Water: US Geological Survey Open-File Report 2009-1004, 8 p.

Kasper, J., and Strohmeier, S. A., 2007, Ground-water-quality survey of the Indian River Bay watershed, Sussex County, Delaware: Results of sampling, 2001-03: Delaware DNREC, 40-08-05/07/03/01, 57 p.

Koerner, E. L., and Haws, D. A., 1979, Long-term effects of land application of domestic wastewater, Vineland, New Jersey, rapid infiltration site: USEPA, EPA600/2-79-072, 167 p.

Koterba, M. T., Andres, A. S., Vrabel, J., Crilley, D. M., Szabo, S., DeWild, J. F., Aiken, G. R., and Reyes-Padro, B., 2006, Occurrence and distribution of Mercury in the surficial aquifer, Long Neck Peninsula, Sussex County, Delaware: U.S. Geological Survey Scientific Investigations Report 2006-5011, 62 p.

- Leach, L. E., Enfield, C. G., Harlin, C. C., 1980, Summary of long-term rapid infiltration system studies: USEPA, EPA600/2-80-165, 51 p.
- LeBlanc, D. R., 1984, Sewage plume in a sand and gravel aquifer: US Geological Survey Water -Supply Paper 2218.
- Lee, R. W., and Bennett, P. C., 1998, Reductive dissolution and reactive solute transport in a sewage-contaminated glacial outwash aquifer: *Ground Water*, vol. 26, p. 583-595.
- Leis, W. M., 1974, Artificial recharge in coastal Sussex County, Delaware: University of Delaware Water Resources Center, 100 p.
- Liu, C., and Ball, W. P., 1999, Application of inverse methods to contaminant source identification from aquitard diffusion profiles at Dover AFB, Delaware: *Water Resources Research*, v. 35, p. 1975-1985.
- Maguire, R. O., and Sims, J. T., 2002, Measuring Agronomic and Environmental Soil Phosphorus Saturation and Predicting Phosphorus Leaching with Mehlich 3: *Soil Science Society of America Journal*, vol. 66, p. 2033-2039.
- McCobb, T. D., LeBlanc, D. R., Walter, D. A., Hess, K. M., Kent, D. B., and Smith, D. B., 2002, Phosphorus in a Ground-Water Contaminant Plume Discharging to Ashumet Pond, Cape Cod, Massachusetts, 1999: US Geological Survey Water-Resources Investigation Report 02-4306, 69 p.
- McCobb, T. D., LeBlanc, D. R., and Massey, A. J., 2009, Monitoring the removal of phosphate from ground water discharging through a pond-bottom permeable reactive barrier: *Ground Water Monitoring and Remediation*, v. 29, p. 43-55.
- McMahon, P.B., and Chapelle, F.H., 2008, Redox processes and water quality of selected principal aquifer systems: *Ground Water*, v. 46, p. 259-271.
- Michalak, A. M., and Kitindis, P. K., 2004, Application of geostatistical inverse modeling to contaminant source identification at Dover AFB, Delaware: *Journal of Hydraulic Research*, v. 42, p. 9-18.
- Miller, J.C., 1972, Nitrate Contamination of the Water-Table Aquifer in Delaware: Delaware Geological Survey Report of Investigations No. 20, 36 p.
- Moura, D. R., 2009, Phosphorus fate and transport in wastewater applied to rapid infiltration basins: M.S. Thesis, University of Florida, 105 p.
- Natural Resource Conservation Service, 2010, Web Soil Survey: <http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm>
- Oliva, J. A., 1985, Operations at the Cedar Creek wastewater reclamation-recharge facilities, Nassau County, New York, in Asano T. ed, *Artificial recharge of ground water*, p. 397-424.
- Oren, O., Gavrieli, I., Burg, A., Guttman, J., and Lazar, B., 2006, Manganese mobilization and enrichment during soil aquifer treatment (SAT) of effluent, the Dan Region Sewage Reclamation Project, Israel: *Environmental Science and Technology*, v. 41, p. 766-772.
- Parkhurst, D. L., Stollenwerk, K. G., and Colman, J. A., 2003, Reactive-transport simulation of phosphorus in the sewage plume at Massachusetts Military Reservation, Cape Cod, Massachusetts: US Geological Survey Water-Resources Investigation Report -03-4017.
- Pellerito, V., Neimester, M.P., Wolff, E., and Andres, A.S., 2006, Results of the Domestic Well Water Quality Study: Delaware Geological Survey Open File Report No. 48, 50 p.

- Pound, C. E., Crites, R. W., and Olson, J. V., 1979, Long-term effects of land application of domestic wastewater, Hollister, California, rapid infiltration site: USEPA, EPA600/2-78-084, 150 p.
- Ramsey, K. W., 2005, Geologic Map of New Castle County, Delaware: Delaware Geological Survey Geologic Map No. 13.
- Ramsey, K. W., 2007, Geologic Map of Kent County, Delaware: Delaware Geological Survey Geologic Map No. 14.
- Repert, D.A., Barber, L.B., Hess, K.M., Keefe, S.H., Kent, D.B., LeBlanc, D.R., and Smith, R.L., 2006, Long-Term Natural Attenuation of Carbon and Nitrogen within a Groundwater Plume after Removal of the Treated Wastewater Source: *Environmental Science & Technology*, v. 40, p. 1154-1162.
- Ritter, W.F., and Chirnside, A.E.M., 1982, Ground Water Quality In Selected Areas of Kent and Sussex Counties, Delaware: Agriculture Experiment Station, University of Delaware.
- Ritter, W.F., and Chirnside, A.E.M., 1984, Impact of Land Use on Ground-Water Quality in Southern Delaware: *Ground Water*, v. 22, p. 38-47.
- Rivett, M. O., Buss, S. R., Morgan, P., Smith, J. W. N., and Bemment, C. D., 2008, Nitrate attenuation in groundwater: A review of biogeochemical controlling processes: *Water Research* v. 42, p. 4215-4232.
- Robertson, F.W., 1977, The quality and potential problems of ground water in coastal Sussex County, Delaware: University of Delaware Water Resources Center, 58 p.
- Robertson, W.D., 2003, Enhanced attenuation of septic system phosphate in noncalcareous sediments. *Ground Water* v. 41, p. 48-56.
- Robertson, W.D., Schiff, S.L., Ptacek, C.J., 1998, Review of phosphate mobility and persistence in 10 septic system plumes. *Ground Water* 36, p. 1000- 1010.
- Robertson, W. D., Harman, J, 1999, Phosphate plume persistence at two decommissioned septic system sites. *Ground Water* v. 37, p. 228-236.
- Robertson, W.D., Ptacek, C.J., and Brown, S. J., 2009, Rates of nitrate and perchlorate removal in a 5-year-old wood particle reactor treating agricultural drainage: *Ground Water Monitoring and Remediation*, v. 29, p. 87-94.
- Sallade, Y.E., and Sims, J.T., 1997, Phosphorus transformations in the sediments of Delaware's agricultural drainageways: II. Effect of reducing conditions on phosphorus release: *Journal of Environmental Quality* v. 26, p. 1579-1588.
- Scheibe, T. D., and Freyberg, D. L., 1995, Use of Sedimentological Information for Geometric Simulation of Natural Porous Media Structure: *Water Resources Research*, v. 31, p. 3259-3270.
- Scheibe, T., and Yabusaki, S., 1998, Scaling of flow and transport behavior in heterogeneous groundwater systems: [Advances in Water Resources](#), v. 22, p. 223-238.
- Schneider, B. J., and Oaksford, E. T., 1986, Design, operation, and monitoring capability of an experimental artificial-recharge facility at East Meadow, Long Island, New York: US Geological Survey Water-Resources Investigation Report 84-4321, 46 p.
- Schneider, B. J., Ku, H. F., and Oaksford, E. T., 1987, Hydrologic effects of artificial-recharge experiments with reclaimed water at East Meadow, Long Island, New York: : US Geological Survey Water-Resources Investigation Report 85-4323, 79 p.

Shah, S., and Fox, P., 2005, Conversion of nitrate to nitrite as a rate limiting step for anaerobic ammonia oxidation in soil systems: Water Environment Federation, WEFTEC 2005, p. 5517-5537.

[Shahid](#), R., and Viraraghavan, T., 1987, Use of peat in septic tank effluent treatment - column studies: Water Pollution Research Journal of Canada, v 22, p. 491-504.

Sims, J. T., Andres, A. S., Denver, J. M., and Gangloff, W. J., 1996, Assessing the impact of agricultural drainage on ground and surface waters in Delaware: University of Delaware Department of Plant and Soil Science, 185 p.

Sims, J. T., Andres, A. S., Denver, J. M., and Gangloff, W. J., Vadas, P. A., and Ware, D. R., 1998, Assessing the impact of agricultural drainage on ground and surface waters in Delaware, Development of best management practices for water quality protection: University of Delaware Department of Plant and Soil Science, 80 p.

Sims, J. T., Maguire, R. O., Leytem, A. B., Gartley, K. L., and Pautler, M. C., 2002, Evaluation of Mehlich 3 as an agri-environmental soil phosphorus test for the Mid-Atlantic United States of America: Soil Science Society of America Journal, vol. 66, p. 2016-2032.

Smith, R. L., Garabedian, S. P., and Brooks, M. H., 1996, Comparison of denitrification activity measurements in groundwater using cores and natural-gradient tracer tests: Environmental Science and Technology, v. 30, p. 3448-3456.

Sparks, D. L., Sims, J. T., Seiter, J., and Gardner, S., 2007, Fate and transport of arsenic in Delaware soils: assessing potential impacts on water quality: University of Delaware Department of Plant and Soil Science, unpublished report submitted to DNREC, 97 p.

Stracek, O., Bhattacharya, P., Jacks, G., Gustafsson, J., and von Bromssen, M., 2004, Behavior of arsenic and geochemical modeling of arsenic enrichment in aqueous environments: Applied Geochemistry, v. 19, p. 169-180.

Stollenwerk, K. G., 2003, Simulation of phosphate transport in sewage-contaminated groundwater, Cape Cod, Massachusetts; Applied Geochemistry, v. 11, p. 317-324.

Sumner, D.M., and Bradner, L.A., 1996, Hydraulic Characteristics and Nutrient Transport and Transformation Beneath a Rapid Infiltration Basin, Reedy Creek Improvement District, Orange County, Florida: U. S. Geological Survey, Report WRI 95-4281, 51 p.

Szabo, Z., dePaul, V.T., Kraemer, T.F., Parsa, B., 2004, Occurrence of radium-224, radium-226, and radium-228 in water of the unconfined Kirkwood-Cohansey aquifer system, southern New Jersey: U.S. Geological Survey Scientific Investigations Report 2004-5224, 92 p.

Tartakovsky, A. M., Meakin, P., Scheibe, T. D., and West, R. M. E., 2007, Simulations of reactive transport and precipitation with smoothed particle hydrodynamics, Journal of Computational Physics, v. 222, p. 654-672.

Turkmen, M., Walther, E. F., Andres, A. S., Chirnside, A. E., and Ritter, W. F., 2008, Evaluation of rapid infiltration basin systems (RIBS) for wastewater disposal: Phase I: Delaware Geological Survey unpublished contract report to DNREC, 41 p.

US Army Corps of Engineers, 1997, Site inspection report – former Fort Miles Military Reservation, Sussex County, Delaware: Baltimore, Maryland, US Army Corps of Engineers Engineering Division.

Ullman, W. J., Andres, A. S., Scudlark, J. R., and Savidge, K. B., 2002. Storm-water and baseflow sampling and analysis in the Delaware Inland Bays: preliminary report of findings 1998-2002: Open-File Report No. 44, Delaware Geological Survey, Newark, DE, 40 p.

United States Environmental Protection Agency, 1985, Process Design Manual for Land Treatment of Municipal Wastewater : Supplement On Rapid Infiltration and Overland Flow: Cincinnati, Ohio, U.S. Environmental Protection Agency, 121 p.

_____, 2004, Guidelines for Water Reuse: U.S. Agency for International Development, U.S. Environmental Protection Agency Report EPA/625/R-04/108, 1-450 p.

_____, 2006, Process Design Manual Land Treatment of Municipal Wastewater Effluents, U.S. Environmental Protection Agency Report EPA/625/R-06/016.

Vadas, P.A., and J.T. Sims, 1998, Redox status, poultry litter, and phosphorus solubility in Atlantic Coastal Plain soils. *Soil Science Society America Journal*, v. 62, p. 1025-1034.

Vadas, P.A., and J.T. Sims, 1999, Phosphorus sorption in manured Atlantic Coastal Plain soils under flooded and drained conditions: *Journal of Environmental Quality*, v. 28, p. 1870-1877.

Vadas, P. A., Srinivasan, M. S., Kleinman, P. J., Schmidt, J. P., Allen, A. L., 2007, Hydrology and groundwater nutrient concentrations in a ditch-drained agroecosystem: *Journal of Soil and Water Conservation*, p. 178-187.

Walter, D. A., Rea, B. A., Stollenwerk, K. G., and Savoie, J., 1995, Geochemical and hydrologic controls on phosphorus transport in a sewage-contaminated sand and gravel aquifer near Ashumet Pond, Cape Cod, Massachusetts: *US Geological Survey Water Supply Paper 2463*, 65 p.

Wood, C., 2006, Wastewater reclamation and groundwater recharge for a sole-source aquifer: Dartmouth College: *Dartmouth Undergraduate Journal of Science*, p. 4-8.

Woodruff, K. D., 1970, General ground-water quality in fresh-water aquifers of Delaware: : Delaware Geological Survey Report of Investigations No. 15, 22 p.

YSI, 2008, YSI 556 MPS operations manual: YSI Environmental, Yellow Springs, OH, 114 p.

Zurawsky, M. A., Robertson, W. D., Ptacek, C. J., and Schiff, S. L., 2004, Geochemical stability of phosphorus below septic system infiltration beds: *Journal of Contaminant Hydrology*, v. 73, p. 129-143.

Zvomuya, F., Rosen, C. J., and Gupta, S. C., 2006, Phosphorus sequestration amendments to reduce leaching from wastewater applications: *Journal of Environmental Quality*, v. 35, p. 207-215.