Hood Canal Onsite Sewage System Nitrogen Loading Project: Year 2 Final Report

Report to the Puget Sound Partnership

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I. Executive Summary

A. General Background

Portions of Hood Canal suffer from intermittently low dissolved oxygen concentrations, which are induced by oxygen consumption associated with decay of phytoplankton. The growth of phytoplankton in the Canal is limited by the availability of nitrogen (N). Although the natural physical and chemical characteristics of the Canal can induce episodes of low dissolved oxygen naturally, anthropogenic N additions may be affecting the nutrient dynamics in Hood Canal. Nearly all residences and businesses in the Hood Canal watershed treat wastewater using conventional onsite sewage systems (OSSs) which are not designed to remove N. Therefore, nitrogen from these systems may contribute to the hypoxic events in the Canal. A critical question is how much watershed N export is required to cause hyoxia in Hood Canal, and can OSS N discharges cause the Hood Canal system to exceed a "tipping point"?

The load of N from OSSs to Hood Canal depends on the number of systems in the watershed, the average rate of N release from these systems, and the extent of N removal from the effluent during migration from the origin of the sewage to the Canal or its related watershed systems. The first two of these factors are easily quantified. Nitrogen removal, however, is variable and difficult to determine. The primary mechanism for N removal is denitrification – the conversion of nitrate to molecular nitrogen. Denitrification is a reduction-oxidation (redox) reaction in which N in a nitrate ion acquires electrons; therefore, an electron donor must be available for the reaction to proceed. In addition, the reaction occurs at a significant rate in the environment only when it is mediated by bacteria that reduce nitrate under anoxic or suboxic conditions. Denitrification has been studied in other locations and in laboratory experiments. However, the applicability of these studies to Hood Canal is limited due to the heterogeneity of conditions influencing denitrification activity. Plant uptake may also influence the fate and transport of OSS N discharges, but it is generally believed plant uptake merely delays the transport of N. However, during the process of plant uptake and release N may be converted to less labile dissolved organic forms.

The overarching goal of the Hood Canal Onsite Sewage System Nitrogen Project was to assess the extent of N removal from OSS effluent occurring in the sub-surface down-gradient of OSSs within the Hood Canal watershed. This study was conducted over two years and characterized nitrogen removal at eight sites.

The first year of the study focused on five sites located at varying distances from Hood Canal. Nitrogen removal appeared to vary greatly among the five sites. Substantial losses of N down-gradient of OSS drainfields were inferred at three of the sites, suggesting that denitrification may have occurred at those sites. Most of the N removal that occurred at those sites appeared to occur very close to the drainfield. At the other two sites, however, the data were most consistent with conservative N transport characterized by little or no loss due to denitrification. However, the conclusions pertaining to those sites were associated with a high level of uncertainty due to a low number of samples, uncertainty concerning the sub-surface migration path of the N, and the fate of the N as it progresses along the path of migration.

The variability in nitrogen removal among the studied sites suggests a wide range of nitrogen removal throughout the Hood Canal watershed. However, experimental complications in the study lent significant uncertainty to the conclusions. The challenges included limited access to groundwater, a limited number of monitoring events, difficulty identifying OSS plumes, and lack of background groundwater for comparison. The goals of the Year 2 study addressed the challenges confronted during Year 1. This study is also constrained by the fact that saturated conditions (which were necessary to collected samples from our shallow wells) were only present during the wetter fall and winter months, therefore most of our sampling was restricted to these periods.

B. Study Approach

The overarching project goal was to quantify the removal of N from OSS effluent in different subsurface environments within the Hood Canal watershed. During Year 2, experiments were conducted at four study sites, designated Mason 5 (MAS05), Mason 7 (MAS07), Mason 10 (MAS10) and Kitsap 3 (KIT03). MAS05 and MAS07 are residential sites located immediately adjacent to the shoreline of southern Hood Canal and are confined by bulkheads. Neither of these sites meet current rules and requirements for design and location. The subsurface geology at both sites consists of glacial till overlain by reworked fill material comprised of beach deposits and outwash. MAS10 is a commercial facility approximately 0.75 km from Hood Canal’s terminus at Lynch Cove, in Belfair. The OSS at MAS10 serves, on average, five to seven employees. The subsurface at MAS10 is mapped as outwash. KIT03 is a residential site located 1.6 km upland from Hood Canal and lies above glacial till. KIT03 was also sampled during Year 1 of this project.

The varying characteristics of study sites required different experimental approaches to achieve the project goals. In all cases, it was necessary to identify a conservative tracer that could be used to distinguish between OSS effluent and other sources that might have contributed constituents to subsurface water samples, specifically groundwater uninfluenced by the OSS and, at the shoreline sites, seawater. Accordingly, efforts were made to sample groundwater down-gradient of OSSs, OSS effluent. and background water unaffected by OSS discharges. The frequency of sample collection, the depth of sampling wells, the conservative tracers used and the experimental details varied among the sites. The shoreline sites (MAS05 and MAS07) were studied in three sampling events during which subsurface water samples were periodically collected from shallow wells over the course of successive tidal cycles. To assess the contribution of seawater to the samples, bromide was used as a conservative tracer. Subsurface water was not accessible via shallow sampling wells at MAS10, so deeper wells penetrating to a saturated zone approximately 6 m below ground surface were installed. The investigation of MAS10 consisted of both routine sampling and artificial rainfall experiments. KIT03 was investigated through routine sampling of perched groundwater accessed via shallow wells. Chloride, which is naturally present in OSS effluent, and bromide, which was added to the septic tank, were used as conservative tracers at KIT03. Domestic drinking water wells were also sampled during Year 2 in order to characterize the shallow groundwater that OSS effluent interacts with in the Hood Canal watershed. After percolating through the vadose zone, OSS effluent may be diluted by ambient groundwater depending upon the location and the season. The characteristics of this groundwater were assessed since they may determine whether or not N reduction is likely to occur in the saturated zone when it is present.

The probabilistic model developed during Year 1 of the study to assess N removal down-gradient of OSS drainfields was used to analyze the Year 2 results. The model differentiates between the dilution and loss of N, taking into account the uncertainty in the N concentrations in OSS effluent and the background water diluting that effluent.

C. Results & Discussion

The extent of N removal from OSS effluents varied considerably amongst the study sites. No significant N removal occurred at the shoreline sites, but some N removal did appear to occur at intermediate and upland sites. The probabilistic model indicated that the median N removal at the shoreline sites was 9%, and that the probability that all N was transported conservatively was 72%. The minimal N removal at these sites was attributed to oxygenation and rapid transport of OSS effluent to the Canal by seawater that infiltrates the drainfield during high tides and then drains back to the Canal. Data indicated that such tidally infiltrating seawater diluted OSS effluent by a factor of 100 to 1000 within 6.5 m of the drainlines at MAS05 and MAS07. The interpretation of the results at MAS10, the intermediate site, was complicated by the possibility that the site may have been influenced by an off-site OSS. Nonetheless, results indicated that partial N removal occurred close to the drainfield. The median level of N removal at MAS10, according to the model calculations, was 37%. The experimental data also supported partial N removal within the drainfield at the upland site, KIT03. The median estimated N removal values during Years 1 and 2 at this site were 66% and 39%, respectively. It was hypothesized that N removal was limited by the availability of electron donors and the presence of oxic conditions at the upland and intermediate sites. The estimated N removal at all the Year 2 sites and at the Year 1 sites where sufficient data were available to draw conclusions are summarized in Table A.

Table A. Site descriptions and median nitrogen removal estimates

Site Information			Median N		Experimental Information				
Site	**Location**	**OSS Type**	**Y1**	**Y2**	**N Source**	**Tracer**	**Background Source**	**Percent Sampling** [3]	**Time Period**
Idealized	*NA*	*NA*	*NA*	*NA*	*OSS*	*Cl^- and Br^-*	*Ambient Groundwater/Seawater*	*100*	*2 Years*
MAS01	Intermediate	Pump[1]	15%	NA	OSS	Cl^-	Tap Water	59	1 Year
MAS02	Intermediate	Mound[2]	92%	NA	OSS	Cl^-	Ambient Groundwater	60	1 Year
MAS05	Shoreline	Gravity	NA	7%	OSS	Br^- Reverse	Seawater	60	1Year
MAS07	Shoreline	Gravity	NA	12%	OSS	Br^- Reverse	Seawater	27	1 Year
MAS10	Intermediate	Gravity	NA	37%	Unsure	Cl^-	Off-site Groundwater	150	1 Year
KIT01	Inland	Gravity	0%	NA	OSS	Cl^-	Tap Water	16	1 Year
KIT02	Intermediate	Gravity	81%	NA	OSS	Cl^-	Tap Water	7	1 Year
KIT03	Inland	Gravity	66%	39%	OSS	Cl^- and Br^-	Ambient Groundwater	88	2 Years

[1] Septic tank pumped to drainfield, activated by float switch

[2] Septic tank pumped to elevated drainfield

[3] Number of samples collected as a percentage of project target

D. Conclusions

This two-year study assessed the probability of N removal in the drainfields of eight OSSs in the Hood Canal watershed. During Year 1 of the study, a consistent method was applied to five sites, however, this method was only minimally effective for collecting water samples at two of the sites. Accordingly, site-specific methods were employed during Year 2. In addition to the results gained from this study, many difficulties associated with achieving the research objectives in this region were identified, and potential ways of addressing these difficulties are suggested.

Over the course of the study, a wide range of N removal was observed at the study sites. The variability observed in experimental results was attributed to the distinct characteristics of each study site. The true variability of N removal within the watershed cannot be defined by a small number of study sites. Like similar studies of denitrification, this field study concludes that the occurrence and extent of denitrification is spatially specific and highly variable. However, the case studies in this report and the Year 1 report (Horowitz et al. 2007) present different scenarios that can inform our understanding of environmental conditions influencing N removal from OSS effluent within the Hood Canal watershed.

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1. Introduction

1.1 Study Overview

The goal of the Hood Canal Onsite Sewage System (OSS) Nitrogen Loading project is to assess the fate of nitrogen between OSSs and site property boundaries within the Hood Canal watershed. The contextual background for the study and results from the first year can be found in the Year 1 Final Report (Horowitz et al. 2007). This report details the findings of the project's second and final year, during which four OSSs were intensively investigated. Routine monitoring and site-specific experiments were conducted to better understand the extent of nitrogen (N) removal occurring down-gradient of several OSSs. Nitrogen loading to OSSs can be quantified based on measurements of inputs, but after water exits the septic tank, subsurface processes can remove N from these discharges. The extent to which these processes occur in the Hood Canal watershed is not known. Denitrification, the biological reduction of nitrate (NO_3) to molecular N_2, is the primary process by which N is removed from OSS effluent. The extent of denitrification occurring in a given subsurface environment depends strongly on site-specific conditions. Nitrogen can also be taken up an temporary retained by vegetation, but it is generally believed the plant uptake only delays the travel time for N and does not removal it from the system (Hedin et al. 1998). However, plant uptake can ultimately convert bioavailable dissolved inorganic N to less bioavailable dissolved organic nitrogen especially in nutrient poor forests (Neff et al. 2003).

To address this issue, four study sites representing three different subsurface environments characteristic of the Hood Canal watershed were investigated. The extent of N removal from OSS effluent was assessed at each site through analysis of samples collected from and down-gradient of the OSS. This study was part of a broader research effort to quantify N loads potentially contributing to the depletion of dissolved oxygen within certain regions of Hood Canal.

1.2 Hood Canal

Hood Canal, a basin within Puget Sound, is a fjord in western Washington separating the Olympic and Kitsap Peninsulas. The canal stretches approximately 97 km southward along nearly 290 km of shoreline. The last 24 km of the canal hooks northeast to its terminus at Lynch

Cove. Hood Canal varies in width from 0.8 km to 6.5 km, and in depth from shallow tidal flats to greater than 150 m (PSAT and HCCC, 2004; Hull and Bryan, 2005). A sill transecting the Canal from east to west at its northerly inlet creates a shallow zone that impedes deep tidal exchange in the canal (PSAT and HCCC, 2004). Hood Canal is an estuarine system receiving marine water through the Strait of Juan de Fuca and freshwater from five major rivers originating in the Olympic Mountains and many smaller streams. These Olympic Mountain rivers are: the Skokomish River, the Duckabush River, the Hamma Hamma River, the Big Quilcene River and the Dosewallips River.

The physical characteristics of Hood Canal make it naturally prone to low dissolved oxygen (DO). The deep, narrow shape of the channel, coupled with a restricted outlet and freshwater inputs, impedes circulation and deep water oxygen resupply. Freshwater inputs lay above colder, saline water, resulting in strong density stratification. This salinity-driven stratification persists due to weak tidal exchange with Puget Sound. The surface water generally flows northward as the deep, saline water flows southward. These complex circulation dynamics make the system susceptible to hypoxia (PSAT and HCCC, 2004). Low DO levels were recorded as early as the 1950s, when hypoxic events were largely confined to the Lynch Cove area and lasted an average of three to six months during most years.

Data collected by the Puget Sound Ambient Monitoring Program (PSAMP) and the Washington State Department of Ecology (DOE) indicate that the periods of low DO are increasing in frequency and intensity (PSAT and HCCC, 2004; Paulson et al., 2006). Researchers hypothesized that eutrophication due to anthropogenic N loading might be a contributing cause of the deteriorating water quality conditions (PSAT and HCCC, 2004).

1.2.1 Nitrogen Loading to Hood Canal

Both natural and anthropogenic inputs provide nitrogen to Hood Canal. The DOE has classified Hood Canal as a water body of "very high concern," characterized by both natural sensitivity to nutrient inputs and a high likelihood that human activities are exacerbating the decline of water quality (PSAT and HCCC, 2004). A preliminary assessment by the United States Geological Survey (USGS) (Paulson et al., 2006) identified marine contributions as the dominant source of N, followed by inputs from rivers and streams, regional groundwater, onsite wastewater treatment systems, atmospheric deposition, and other sources. Nitrogen inputs to Hood Canal from deep seawater exceed the human-derived N contribution. However, of all the N entering the Canal from seawater, only the N in the water that upwells into the euphotic zone contributes to eutrophication. The additional N load attributed to humans could potentially contribute to the increased frequency and severity of hypoxic events in Hood Canal (PSAT and HCCC, 2004).

1.2.2 Onsite Sewage Systems and Hood Canal

Hood Canal's approximately 54,000 residents reside in an estimated 24,800 housing units (PSAT and HCCC, 2004). A recent survey of 881 homeowners living within the watershed indicated that 86% of the households use either conventional or pressurized OSSs, consisting of a septic tank and drainfield (Christensen, 2005). In addition to this, wastewater from an unquantified number of commercial and public facilities is discharged to OSSs.

The Preliminary Assessment and Corrective Action Plan (PACA) (PSAT and HCCC, 2004) identified OSSs as an important non-point source of anthropogenic N loading to Hood Canal. The report estimated that residential OSSs annually contribute between 35 and 219 MT of N to Hood Canal, accounting for 33% to 84% of the estimated total anthropogenic N loading. These estimates were based on simplified best- and worst-case scenarios. A subsequent study conducted by the USGS estimated the N load originating from shallow OSSs along the Hood Canal shoreline to be 26 ± 15 metric tons (MT) annually (Paulson et al., 2006). However, this calculation assumed an aggregate 35% reduction in the total dissolved nitrogen (TDN) load due to retention and removal, based on data in literature unrelated to Hood Canal. The actual N removal from such effluent along Hood Canal could differ substantially from this assumed value. The Puget Sound Partnership (PSP) therefore concluded that research on N removal from OSS effluent plumes within the Hood Canal watershed could provide valuable data for assessing the potential N load from OSSs to Hood Canal.

1.3 Review of Year 1

The goal during Year 1 of the Hood Canal Onsite Sewage System Nitrogen Loading study was to estimate nitrogen removal within 100 feet down-gradient of sewage systems at study sites in the Hood Canal watershed. A summary of the conclusions from the Year 1 report are described in this section, as are the challenges confronted during Year 1 which directed the research focus during Year 2.

1.3.1 Conclusions from Year 1

The Year 1 study focused on five spatially and geologically diverse sites in the Hood Canal watershed. Sites Kitsap 1 and Kitsap 3 are located near Seabeck at upland locations in glacial till substrate. Kitsap 2, near Silverdale and adjacent to Hood Canal, is at a transition zone between Vashon glacial till and undifferentiated glacial deposits (Yount, 1993). Mason 1 is adjacent to Hood Canal, on the south side of East State Route 106 in a coarse gravel with sand, silt, and clay that underlies Vashon till (Molenaar and Noble, 1970). . Mason 2 is on the floodplain of the Skokomish River, approximately 1.5 miles from Hood Canal. It is located on fine-grained alluvial soils deposited by the Skokomish River (Molenaar and Noble, 1970).

Shallow groundwater sampling wells were installed in and around the drainfield at each of these study sites in the fall and winter of 2006. Well samples, collected throughout the subsequent winter and spring, were analyzed for a suite of constituents including ammonium, nitrate, total dissolved nitrogen, dissolved organic carbon, and chloride. Chloride (Cl^-) was used as a conservative tracer of OSS effluent. A probabilistic model was developed to quantify N removal by comparing changes in the concentrations of total N and Cl^- downstream of the septic tanks. N removal was estimated at each study site.

The inferred extent of nitrogen removal varied greatly among the five sites. Substantial losses of N down-gradient of the OSSs at the Mason 2 and Kitsap 3 sites suggested that denitrification was occurring at those sites, and moderate N removal seemed to be occurring at the Kitsap 2 site. N transport was largely conservative at the Mason 1 and Kitsap 1 sites. The variability in N removal among the sites suggested wide ranges of N removal from OSS discharges throughout the Hood Canal watershed. It was hypothesized that this variability might reflect differences in the organic and hydraulic loadings, effluent travel distance to saturated conditions, soil texture, soil organic carbon, and perhaps several other parameters affecting the potential for denitrification in the subsurface.

1.3.2 Challenges during Year 1

Several challenges in studying OSS effluent plumes in the Hood Canal watershed were encountered during the first year of this study. Several sites initially intended for inclusion in the study could not be used for the following reasons:

1) inability to draw water out of sampling wells at certain sites because wells did not penetrate a saturated zone
2) difficulty identifying the OSS plume at other sites due to extensive dilution by groundwater
3) limited ability to sample at certain sites because wells only contained water during wet weather conditions
4) mixing of OSS effluent plumes with marine water from Hood Canal which overwhelmed the OSS effluent chloride signal
5) inability to measure the fate of N at further distances from the drainfields
6) limited number of monitoring events
7) limited seasonality of monitoring events

These challenges motivated new questions regarding the fate of OSS effluent, including:

1) How much N removal occurs at sites where wells did not penetrate a saturated zone?
2) Is N stored in the subsurface around drainfields? If so, when is it mobilized?
3) What is the fate of OSS N in shoreline systems?
4) Are the OSS effluent plumes being captured using the Year 1 sampling methods?

The second year of this study attempted to address these questions using new methods. It was realized that using specific methods at different sites would yield more data than continuing to use a consistent method at all sites. The goals and the methods used to address the questions are described in the following section.

1.4 Year 2 Objectives

The overarching goal of the second year of the Hood Canal OSS Nitrogen Loading study was the same as the first year: to estimate nitrogen removal down-gradient of OSSs at study sites in the Hood Canal watershed. However, research conducted during Year 2 specifically addressed the challenges confronted during Year 1. The study design was altered from a uniform approach at all sites to the use of site-specific approaches. The Year 2 objectives and an overview of the methods used to address these objectives are as follows.

Objective 1) Determine the location of an OSS effluent plume and the extent of N removal at a site where shallow wells might not penetrate a saturated zone.

Deep groundwater wells (6 meters) were installed in Spring 2008 at the Mason 10 field site in Belfair. Bromide (Br^-) was added as a secondary tracer to the septic tank for plume identification. Frequent sampling was conducted following well installation.

Objective 2) Determine N storage and mobilization in and around an OSS drainfield.

Five artificial rainfall experiments were conducted at Mason 10. Four of the experiments were conducted during the fall and winter of 2007. The fifth experiment was conducted in June 2008. The repetition of the experiments allowed for study of seasonal changes in nutrient storage and mobilization.

Objective 3) Determine the fate of OSS N at shoreline sites with tidally influenced drainfields.

Two sites – Mason 5 and Mason 7 – located on the north shore of lower Hood Canal, were studied using tracers and additional analytical methods to identify OSS plumes and assess N removal.

Objective 4) Corroborate the identification of an OSS effluent plume by using an additional conservative tracer.

Kitsap 3, a study site during Year 1, was sampled using the same methods used during the first year of the study. However, Br^- tracer was added to the septic tank at Kitsap 3 during February and March 2008.

Objective 5) Conduct a regional assessment of dissolved organic carbon and dissolved oxygen concentrations in shallow groundwater within the Hood Canal watershed.

> Water samples were collected from shallow residential drinking water wells throughout the Hood Canal watershed and analyzed for dissolved organic carbon and dissolved oxygen as well as other parameters.

1.6 Report Structure

The main body of this report is composed of three sub-sections, each focused on a different study site. Although unique approaches and methods were employed at different sites, all work performed was consistent with the overarching goal of assessing the fate of N discharged from OSSs and estimating the extent of denitrification occurring in the subsurface. All work was also consistent with the project QAPP dated December 11th, 2007. Each sub-section contains the site-specific study goals and approach, site description, methods, results, discussion and conclusions.

2. Methods Common to All Sites

2.1 Shallow Well Installation

Most shallow sampling wells installed during this project consisted of 1.27 cm (0.5 inch) diameter, Schedule 40 PVC pipe plugged at the base, and with approximately 64 holes drilled in the bottom 20 cm of the pipe. Similar wells with 2.54 cm (1 inch) diameters were used to accommodate pressure transducers. Wells at KIT03 were installed by driving a 1.27 cm diameter steel drive point vertically into the ground using a slide hammer. The drive point used at MAS10 was a 3.81 cm (1.25 inch) diameter Geoprobe®. Both these types of drive points were also used at MAS05 and MAS07. After driving, the drive point was removed, and PVC pipe, with 64 holes drilled in the bottom 20 cm of the pipe, was inserted into the hole. Silica sand was poured in the annular space between the well and surrounding soil to create a filter. The sand filled approximately the bottom 25 cm of the space. Bentonite chips were poured over the sand to prevent surface water from preferentially flowing downward along the outside of the PVC pipe into the well. The pipes were then cut off at or above ground surface and capped with PVC slip caps.

A slightly modified approach was also used at MAS05 due to initial difficulties with well installation. During early installation attempts, pea gravel filled in the hole immediately after the drive point was removed. Therefore, the seven wells installed at MAS05 on December 20, 2006 were constructed using 1.27 cm diameter Schedule 80 stainless steel pipes fashioned into drive points. Similar to the PVC pipes, the steel pipes contained approximately 64 holes drilled in the bottom 20 cm of the pipe. The 1.8 m long pipes were driven into the ground using a slide hammer and left there to serve as sampling wells. The tops of these wells were sealed with stainless steel caps.

2.2 Well Sampling

Wells were sampled through dedicated 3/16-inch (0.48-cm) diameter HDPE tubing connected to battery-powered peristaltic pumps. If the recharge rate at a given well allowed, at least three well volumes of water were purged through the sample line before collecting samples. In some cases, wells ran dry after pumping less than one well volume of water, so only the water that was standing in the well at the time of sampling could be collected.

Samples were collected into acid-washed, labeled 250 mL HDPE bottles and stored in a cooler on ice for transport to the laboratory. The bottles were pre-cleaned in the laboratory using deionized water and 0.1 N sulfuric acid (H_2SO_4). If enough water could be collected, the bottles were completely filled in order to minimize headspace. All field samples were labeled with the site, sample ID, date and time.

To check the reliability of the sample handling procedures, duplicate samples were collected for approximately 10% of the 582 total samples by simultaneously pumping a well sample into two labeled 250 mL HDPE bottles. Field blank samples consisting of deionized water were prepared at the start of sampling events to be stored and processed in the same manner as field samples.

2.3 Septic Tank and Blank Sampling

Septic tank samples at KIT03 were collected by attaching a clean, uncapped and labeled sample bottle to a 12.7 mm PVC pipe, and lowering the pipe into the septic tank through the access port until the bottle was submerged. The bottle neck was 1.8 cm (0.7 inch) in diameter, which assured that water entered the bottle slowly. As a result, little water entered the bottle as it was being lowered, and the majority of the sample volume entered when the bottle was several cm below the liquid surface, in the clear water zone of the tank. After capping and rinsing, the samples were double-bagged in plastic Ziploc® bags and placed in the ice-filled cooler for transport to the laboratory.

Septic tank samples at MAS05 and MAS07 were collected using a battery-powered peristaltic pump. A dedicated sample line was submerged below the scum layer of the septic tank, and approximately 200 mL of water was collected from the clear water zone. After capping and rinsing, the bottles were sealed in plastic Ziploc® bags and placed in an ice-filled cooler for transport to the laboratory.

Septic tank samples at MAS10 were collected using the bottle method prior to March 31, 2008, and with the peristaltic pump method after that date.

A blank sample was also placed in the sample cooler to assess the possibility of cross-contamination of samples during handling. This sample consisted of a clean sample bottle filled with deionized water at the laboratory and kept in the sample cooler throughout the field day.

2.4 Pressure Transducers

Pressure transducers were installed in select wells at MAS05, MAS07, and MAS10 in order to continuously record fluctuations in standing water levels at those sites. The units used were Levelogger Gold Model 3001 manufactured by Solinst®. Pressure transducers were installed in wells by suspending the units just above the base of the well with string. Barometers made by Solinst were placed at each study site and were used to compensate for atmospheric pressure. Pressure transducers and barometers recorded pressure values every 15 minutes. Data was routinely extracted from pressure transducers during site visits.

2.5 Field Analyses

At KIT03 and MAS10, 40 mL of well water was collected in a clean 50-ml glass beaker for measurement of temperature and dissolved oxygen using a Yellow Springs Instrument® model 5905 probe connected to a YSI 5000 dissolved oxygen meter. pH and conductivity were measured using Oakton® pH and conductivity probes. All instruments were calibrated at the beginning of each field day, and any discrepancies during calibration were recorded. To avoid contamination of measurement devices and cross-contamination of samples, no field analyses were conducted on septic tank samples. No field analyses were performed at MAS05 and MAS07 because the probes were not suitable for seawater use.

2.6 Laboratory Analyses

Approximately 90 mL of each sample was filtered using plastic laboratory syringes and Whatman® GF/F 0.7 µm glass-fiber filters. Filtrate was collected in clean, labeled 125-mL HDPE bottles and stored in a refrigerator until analysis, which was typically performed within two days of sample collection.

Filtered samples from KIT03 and MAS10 were analyzed for NO_3+NO_2, NH_4^+, Cl^-, Br^-, TDN, and DOC. Filtered samples from MAS05 and MAS07 were analyzed for NO_3+NO_2, NH_4^+, Li^+, Br^- and salinity. Select samples from MAS05 and MAS07 were also outsourced for TDN and DOC analyses. The samples collected during the second experiment at MAS05 were also analyzed for iodide (I^-). The instruments and methods used for laboratory analyses are provided in Table 1.

Table 1. Instruments and methods used for laboratory analyses

Sample Site	Sample Type	Parameter							
		(NO_3+NO_2)-N	(NH_4+NH_3)-N	TDN	DOC	Cl^-	Br^-	I^-	Li^+
		Instrument, Method							
KIT03 & MAS10	Well, Seawater, Blank	AutoAnalyzer, SM 4500-NO_3F	AutoAnalyzer, SM 4500-NH_3G	TN Analyzer1, SM 4500-NB	NPOC Analyzer, EPA 415.1	IC, EPA 300	ICP-MS, EPA 6020	ICP-MS, EPA 6020	ICP-MS, EPA 6020
MAS05 & MAS07	Well, Blank	AutoAnalyzer, SM 4500-NO_3H	AutoAnalyzer, SM 4500-NH_3G	TN Analyzer1, SM 4500-NB	NPOC Analyzer, EPA 415.1	IC, EPA 300	ICP-MS, EPA 6020	N/A	N/A
All Sites	Septic	AutoAnalyzer, SM 4500-NO_3H	AutoAnalyzer, SM 4500-NH_3G	TN Analyzer1, SM 4500-NB	NPOC Analyzer, EPA 415.1	IC, EPA 300	ICP-MS, EPA 6020	ICP-MS, EPA 6020	ICP-MS, EPA 6020

SM	Standard Methods (1998)
AutoAnalyzer	Seal® Model AA3 AutoAnalyzer
TN	Shimadzu Model TOC-VCSH Total Nitrogen Analyzer
NPOC	Shimadzu Model TOC-VCSH
IC	Dionex Ion Chromatograph DX-120
ICP-MS	PerkinElmer Elan DRC-e

3. Shoreline Sites

3.1 Introduction and Goals

The study reported here focuses on the fate and transport of N emanating from OSSs located at properties immediately adjacent to the Hood Canal shoreline. OSS N inputs are easily estimated, but after exiting the septic tank, subsurface processes can remove N from wastewater, and the extent to which these processes occur in the subsurface immediately up-gradient bordering the Hood Canal shoreline is not known. Denitrification represents the dominant N removal process. However, the extent of denitrification occurring in a given environment depends strongly on site-specific conditions which may or may not be met in the subsurface environment adjacent to Hood Canal. The questions guiding this research were:

1) To what extent is N in OSS effluent removed via denitrification as effluent released from near-shore OSSs travels into Hood Canal?

2) What factors limit N removal from OSS discharges in the subsurface environment immediately adjacent Hood Canal?

Routine monitoring as well as targeted experiments were conducted at two shoreline sites to assess the extent of denitrification occurring down-gradient of OSSs adjacent to Hood Canal. The study was based on the concept that N loss could be assessed in the shallow, lateral flow of near-shore OSS discharges though the use of a conservative tracer.

3.2 Shoreline Site Descriptions

The sites investigated in this study are identified as Mason 5 and Mason 7. Both sites are representative of shoreline OSSs along southern Hood Canal. The approximate locations of Mason 5 and Mason 7 are displayed in Figure 1. The parcels are narrow areas extending back from the shore. Like most shoreline residences, the near-shore side of the property is confined by a bulkhead. Both residences are occupied seasonally. In both cases, household wastewater is treated by a shallow, gravity-fed OSS located relatively close to a shoreline bulkhead.

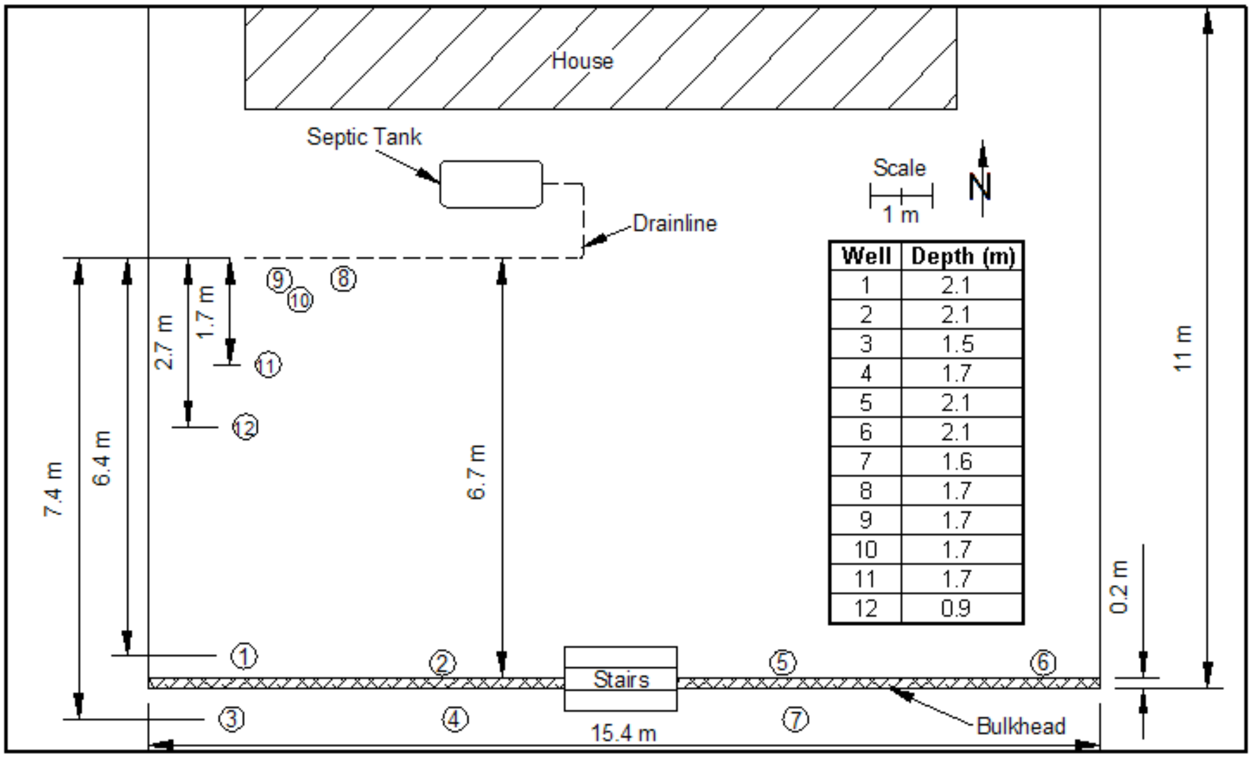
Figure 1. Locations of study sites Mason 5 and Mason 7 (Google Maps)

3.2.1 Mason 5

The Mason 5 site, hereafter referred to as MAS05, is a 645 m^2 residential site located along the Hood Canal shoreline. The site slopes gently southward to the canal, and groundwater is assumed to flow in that direction. The property is elevated approximately 1.2 m above the shoreline. The subsurface geology consists of glacial till overlain by reworked fill material comprised of beach deposits and outwash. There is pea gravel fill near the surface and at greater depths towards the bulkhead. A concrete bulkhead forms the near-shore property boundary. A 170-m^2 single-family household is located approximately 10.5 m behind the bulkhead. Two individuals occupy the residence nine consecutive months each year.

The residence is served by a conventional, gravity-fed OSS installed in 1979. The septic tank has a volume of 1125 gallons (4260 L). The septic tank is installed 2 m down-gradient of the house and 8.5 m up-gradient of the bulkhead. Wastewater exiting the septic tank is conveyed to a single 3-m drainline buried 45 cm and located approximately 6.7 m from the bulkhead.

Seven groundwater sampling wells were installed at MAS05 on Dec. 20, 2006; four wells behind the bulkhead, and three wells on the opposite side below the bulkhead in the intertidal zone. The approximate depth of the wells located behind the bulkhead was 1.8 m. Wells placed below the bulkhead were installed to an estimated depth of 1.25 m. A second set of five wells was installed on May 7, 2008, to an approximate depth of 1.7 m. Two of these wells were sized to accommodate pressure transducers. A site map of MAS05 in addition to measured well depths are shown in Figure 2.



Well	Depth (m)
1	2.1
2	2.1
3	1.5
4	1.7
5	2.1
6	2.1
7	1.6
8	1.7
9	1.7
10	1.7
11	1.7
12	0.9

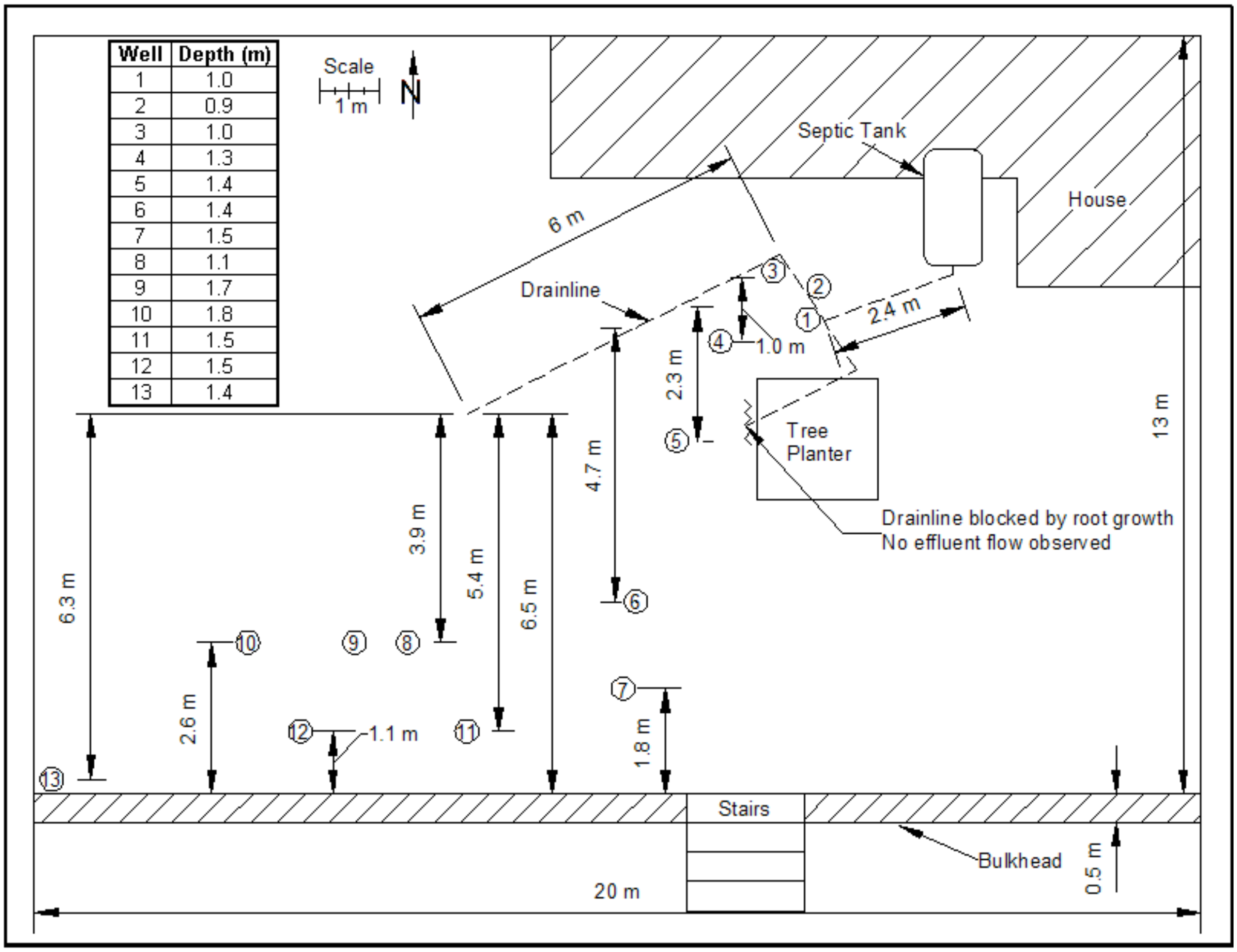
Figure 2. Site map of MAS05 and tabulated well depths

3.2.2 Mason 7

The Mason 7 site, henceforth referred to as MAS07, is an 840 m^2 residential site adjacent to the Hood Canal shoreline. The site slopes slightly south towards the canal, and groundwater is assumed to flow in that direction. The property is elevated about 2.6 m above the shoreline. The subsurface geology consists of glacial till overlain by reworked fill material comprised of beach deposits and outwash. There is 5 cm of gravel fill at the surface. A bulkhead constructed of large rocks forms the near-shore property border. An 83-m^2 single-family household is located approximately 13.5 m behind the bulkhead. Two individuals occupy the residence nine consecutive months each year.

Household wastewater is treated by a conventional, gravity-fed OSS installed in 1949. A concrete septic tank of unknown volume is sited 1.5 m down-gradient of the house and 12 m up-gradient of the bulkhead. Wastewater exiting the septic tank is discharged to a 2.5-m conveyance pipe. The conveyance pipe meets a T-fitting and distributes flow to two drainlines of unknown length angled approximately 27° away from the house. The drainlines are formed by 1-m sections of 10-cm diameter concrete pipe spaced 1.25 cm apart. The drainline nearest the house appears to be functioning properly but the one nearest the bulkhead is blocked by root growth.

Four groundwater sampling wells were installed at MAS07 on Nov. 16, 2006 behind the bulkhead to an approximate depth of 1.8 m. Three additional wells were installed on Dec. 14, 2007, to an average depth of 1.4 m. Six more wells were installed on May 19, 2008 to various depths. A site map of MAS07 and measured well depths are provided in Figure 3.



Well	Depth (m)
1	1.0
2	0.9
3	1.0
4	1.3
5	1.4
6	1.4
7	1.5
8	1.1
9	1.7
10	1.8
11	1.5
12	1.5
13	1.4

Figure 3. Site map of MAS07 and tabulated well depths

3.3 MAS05 and MA07 Site-Specific Methods

3.3.1 Tracer Additions

The initial experimental plan called for the use of lithium as a conservative tracer that could be used to distinguish the OSS signal in well samples. Bromide was not added as a tracer due to high background concentrations in seawater. A concentrated solution of lithium chloride (LiCl) was steadily added directly to septic tanks prior to sampling events. The tracer solution consisted of 20 L of deionized water and 2 kg of LiCl, corresponding to a Li^+ concentration of 16.4 g/L. A peristaltic pump was used to drip the solution into the septic tank. The pump outlet tube was suspended above the water surface to avoid clogging the tubing via microbial growth. The flowrate of the solution was approximately 1.87 L/day. Tracer addition was initiated a minimum of ten days prior to field experiments. The 10-day waiting period exceeded the hydraulic retention time (HRT) of the septic tank, so the effluent discharged just prior to and during the sampling event was enriched with Li^+.

An automated sampling unit was used to collect septic tank samples during the tracer addition phase. The sample line was submerged below the scum layer of the septic tank and collected approximately 200 mL of water from the clear water zone at 6:00 and 18:00 daily during the ten days preceding a sampling event. These samples were analyzed to verify that Li^+ concentrations within the septic tank had stabilized.

3.3.2 Background and Septic Tank Sampling

Approximately 20 seawater samples were collected during the experiments. Samples were collected approximately 2 m beyond the shoreline at various depths in front of the bulkhead. Samples were collected by fully submerging bottles 0.5 m below the water surface. Seawater samples were not collected in areas where surface shoreline seeps were visible. Select seawater samples were measured for DO in the field using a YSI 5905 probe connected to a YSI 5000 dissolved oxygen meter.

3.4 Shoreline Sites Model Overview

A probabilistic model was developed to assess N removal occurring in the subsurface underlying and down-gradient of each OSS drainfield (Horowitz et al. 2007). This model was used in conjunction with a mixing model based on bromide concentrations, which are naturally enriched in seawater. An attempt was made to account for the uncertainty associated with variation in OSS loading due to temporal variations in both flow and composition of the wastewater in addition to the natural variability characterizing seawater.

The model calculations produce values for the expected N concentration in the OSS effluent plumes for any given Br^- concentration, assuming that Br^- is transported conservatively. The N concentrations measured in well samples are then compared to the hypothetical N concentrations that would exist if N was transported conservatively, taking into account the uncertainty characterizing N concentration in OSS effluent and seawater. This comparison of actual and hypothetical values is used to estimate the amount of N removal that occurred in samples. A nearly identical model, employing conductivity instead of Br^- as the conservative tracer, was developed by Weiskel and Howes (1992).

3.4.1 Model Fundamentals

The model is based on a mass balance of a conservative substance in the subsurface (ss). The model assumes that a water sample collected from any well down-gradient of the drainline contains a mixture of septic tank water (sep) and infiltrating seawater (sw). The fraction of seawater (f_{SW}) in each sample can be calculated according to the following:

$$C_{ss}Q_{ss} = Q_{sep}C_{sep} + Q_{sw}C_{sw}$$

$$Q_{sw}C_{sw} = C_{ss}Q_{ss} - Q_{sep}C_{sep}$$

$$Q_{sw}C_{sw} = C_{ss}(Q_{sep} + Q_{sw}) - Q_{sep}C_{sep} \quad \text{(Equation 1)}$$

where C is concentration and Q is flow rate. Dividing through by (Qsep+Qsw), defining the ratio of Qsw/(Qsep+Qsw) as f_{SW}, and solving for f_{SW}, results in:

$$\frac{Q_{sw}}{Q_{sep} + Q_{sw}} C_{sw} = C_{ss} - \frac{Q_{sep}}{Q_{sep} + Q_{sw}} C_{sep}$$

$$f_{sw}C_{sw} = C_{ss} - (1 - f_{sw})C_{sep}$$

$$f_{sw}C_{sw} - C_{sep}f_{sw} = C_{ss} - C_{sep}$$

$$f_{sw}(C_{sw} - C_{sep}) = C_{ss} - C_{sep}$$

$$f_{sw} = \frac{C_{ss} - C_{sep}}{C_{sw} - C_{sep}} \quad \text{(Equation 2)}$$

Assuming that a water sample collected down-gradient of the drainline is composed of seawater and septic tank water, the fraction of OSS effluent in each sample can be computed as follows:

$$f_{ss} = f_{sw} + f_{sep}$$

$$f_{sep} = 1 - f_{sw} \quad \text{(Equation 3)}$$

Equation 2 applies to any conservative substance in the system, such as Br^-. Since field data are available for Css, Csw, and Csep for Br^-, f_{SW} and subsequently, f_{SEP} can be calculated for any sample collected from a well.

The next step involves calculating the expected value of Css for DIN, assuming DIN is conservative. The value is computed using Equation 1. Designating this hypothetical concentration of DIN in the drainfield as Css,conserv,DIN yields:

$$C_{ss,conserv,DIN} = f_{sep} C_{sep,DIN} + f_{sw} C_{sw,DIN}$$

$$C_{ss,conserv,DIN} = \left(1 - \frac{C_{ss,Br} - C_{sep,Br}}{C_{sw,Br} - C_{sep,Br}}\right) C_{sep,DIN} + \left(\frac{C_{ss,Br} - C_{sep,Br}}{C_{sw,Br} - C_{sep,Br}}\right) C_{sw,DIN} \qquad \text{(Equation 4)}$$

The DIN removal in the drainfield is computed by comparing the hypothetical Css,conserv,DIN with the concentration measured in a sample, Css,meas,DIN. The DIN removal percentage for a given DIN concentration is calculated as:

$$DIN_{removal\%} = \left(1 - \frac{C_{ss,meas,DIN}}{C_{ss,conserv,DIN}}\right) \times 100 \qquad \text{(Equation 5)}$$

Alternatively, the equation can be rearranged to indicate the expected value of DIN in the sample, Css,exp,DIN, for any specified percentage removal:

$$C_{ss,\exp,DIN} = C_{ss,conserv,DIN} \left(1 - \frac{DIN_{removal\%}}{100}\right)$$

$$= \left[f_{sep} C_{sep,DIN} + f_{sw} C_{sw,DIN}\right]\left(1 - \frac{DIN_{removal\%}}{100}\right) \qquad \text{(Equation 6)}$$

A visual example of the conservative mixing model is displayed in Figure 4. The conservative mixing member represents the expected DIN concentrations for corresponding Br^- concentrations. Background seawater and septic tank samples represent the upper and lower endpoints of the conservative mixing member. The hypothetical data points A, B, and C represent actual Br^- and DIN concentrations measured in well samples. Data point A indicates conservative transport of N since it falls on the conservative mixing member. Data point B, located below the mixing member, represents N loss because the measured DIN concentration is less than the expected DIN concentration. In some instances, data points fall above the conservative mixing member, as is the case with data point C. Such instances produce a negative value for the apparent N loss and make it appear as if N was produced. Since N fixation or other external sources of N were not expected to influence the samples, all apparent N production is attributed to uncertainty in the model input data.

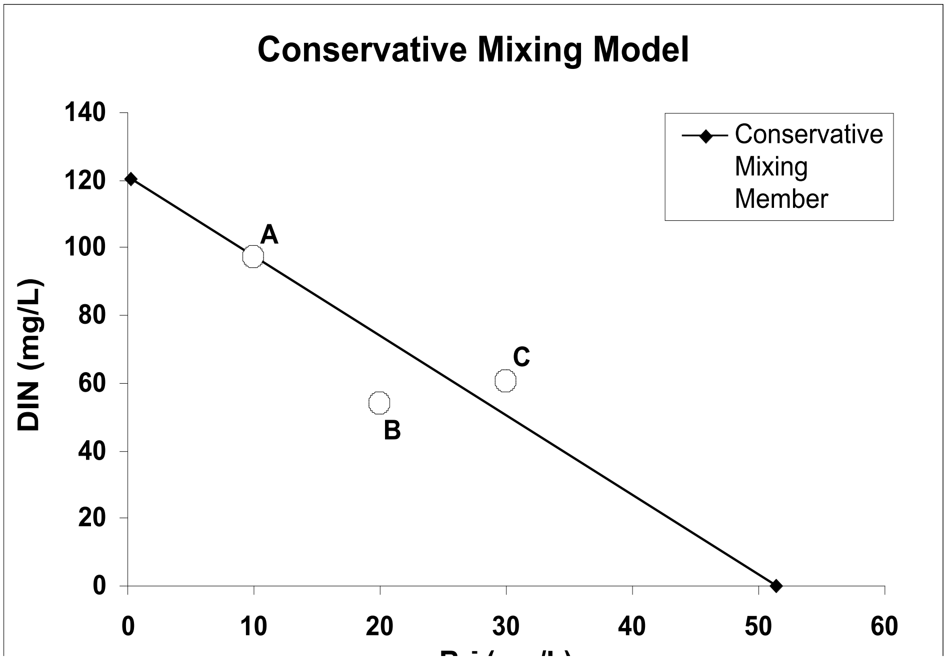


Figure 4. A hypothetical example of the Br^{-}-based conservative mixing model. Point A represents conservative transport of N. Point B represents N removal. Point C represents an error attributed to the variability of model input data.

3.4.2 Probabilistic Modeling

The calculation of $DIN_{removal\%}$ would be simple if the concentrations of Br^{-} and DIN in the seawater and septic tank samples collected at a given site remained constant. However, those values varied between and even within experiments. To account for these variations, Crystal Ball® software (Oracle Corp.), a Microsoft Excel® macro, was used to perform Monte Carlo simulations. In each simulation, 10,000 trials were evaluated using unique values for Csw,Br, Csw,DIN, Csep,Br, and Csep,DIN based on their respective means and standard deviations. During each trial, f_{SEP} was calculated for every measured value of Css,Br, f_{SEP} using Equations 2 and 3. A filter was applied to constrain f_{SEP} to realistic values between zero and one. A value of Css,conserv,,DIN was then calculated in every trial using Equation 6. These calculations produced a set of 10,000 plausible values of Css,conserv,DIN that incorporated the uncertainty in the measured concentrations in the septic tank and seawater. These simulated, plausible values of Css,conserv,DIN are designated Css,sim,DIN.

Next, for each of the 10,000 values of Css,sim,DIN applicable to a given groundwater sample (i.e., calculated using the value of Css,Br in that sample), the percentage of DIN removal required to yield the actual, measured value of Css,DIN in that sample was computed by analogy with Equation 5:

$$DIN_{removal\%,sim} = \left(1 - \frac{C_{ss,meas,TDN}}{C_{ss,sim,TDN}}\right) \times 100 \qquad \text{(Equation 7)}$$

This calculation yields the DIN removal percentage that would have had to occur to yield the DIN concentration measured in the sample, if the randomly generated concentrations in the septic tank and seawater actually applied. Negative values for DIN removal percentages were set equal to zero. Consequently, using the mean and standard deviation of N removal values to characterize model output is not appropriate because the mean of all N removal values is artificially inflated. Instead, the median removal values for wells were calculated and averaged to obtain the average of medians and the standard deviation of those medians. The 10,000 estimated values for the DIN removal percentage in a given sample, corresponding to the full range of probable values for the four uncertain parameters, could be analyzed to obtain a comprehensive assessment of the probable DIN removal percentage in that sample. Similarly, the results for all the samples at a given site could be aggregated to obtain a comprehensive assessment of the DIN removal percentage at the particular site.

3.5 Shoreline Sites Results

3.5.1 Lithium Sorption to Soil

Initially, lithium was intended to serve as the conservative tracer for tracking the OSS plume. Lithium was selected because large amounts of Li^+ could be added to the septic tank, so that the OSS influence on samples could be easily quantified even at high dilutions. Concentrated Li^+ solutions were added to septic tanks in accordance with the methods outlined in Section 5.2. However, substantial decreases in Li^+ concentrations between the septic tanks and wells adjacent to the drainlines suggested that Li^+ sorbed to soil in the drainfield. An experiment was conducted to test this hypothesis.

On August 8, 2008, two 1.5-m holes were dug approximately 3 m up-gradient of the septic tank at MAS05, on opposite sides of the property. Two soil samples of approximately 1 kg each were collected from each hole. A septic tank sample was also collected. Upon returning to the lab, each soil sample was passed through a sieve that allowed only particles <84 μm in diameter to pass. Several 10-g samples of these particles were collected. An unfiltered sample of septic tank water was spiked with 50 mg Li$^+$/L, and 10 mL of this solution was added to each soil sample. Two 10-mL samples of the spiked septic tank solution were used as controls. The ten samples (eight experimental, two control) were shaken for approximately thirty minutes and then incubated at 20 °C for sixteen hours. All samples were then filtered and analyzed for Li$^+$. Experimental results are displayed in Table 2.

Table 2: Experimental results indicating the sorption of Li$^+$ to soil

Li$^+$ Soil Sorption Experiment Results

Sample ID	Sample Description	Li$^+$ Conc (mg/L)	_Li$^+$ (mg/L)	_Li$^+$ (%)
A1	Experimental (Hole 1)	23.4	-29.3	-55.6
A2		20.0	-32.7	-62.0
B1		29.0	-23.7	-45.0
B2		23.4	-29.3	-55.6
C1	Experimental (Hole 2)	21.2	-31.6	-59.9
C2		25.9	-26.8	-50.8
D1		21.8	-30.9	-58.7
D2		22.6	-30.1	-57.1
E1	Control	51.8	n/a	n/a
E2		53.6	n/a	n/a
		Avg.	-29.3	-55.6
		STDEV	2.85	5.41

Exposure of the spiked septic tank solution to soil resulted in approximately a 50–60% loss of Li$^+$. The consistent experimental results demonstrate that Li$^+$ sorbed to the substrate present at MAS05, and presumably to the similar substrate present at MAS07. This interaction between Li$^+$ and the substrate prevented the use of Li$^+$ as a conservative tracer. Hence, an alternative mixing model utilizing the bromide naturally present in seawater was developed to track the OSS plume. The equations and assumptions governing this model were detailed in Section 6.

3.5.2 Pressure Transducer Data

3.5.2.1 MAS05

Pressure transducer data were continuously collected from Well 9 (W9), located adjacent to the drainline at MAS05, between May 9 and July 23, 2008. Those data most relevant to experimental results (i.e., May 22 to June 11) are presented in Figure 5. The water level in W9 ranged from a baseline of 0 cm to 50 cm. Elevated water levels were associated with high tides. Peak water levels typically occurred at or shortly after high tide. On most days, the pattern of water levels mimicked the tidal pattern. Such a pattern was evident in the data from June 3 and 4, as seen in

Figure 5. The influence of the tidal cycle on the water level in W9, indicates that seawater infiltrates past the bulkhead and impacts the subsurface down-gradient of the OSS. However, high tides did not always result in elevated water levels. Although the peak water levels between June 8 and June 10 typically corresponded to high tides, the peak values were substantially lower than those measured in W9 after high tides on June 6 and 7. Therefore, although the tidal cycle influences the water level in W9, other factors exert an influence as well.

The data in Figure 5 are particularly useful for assessing the influence of OSS usage on the water level in W9. The site residents were on vacation from May 22 to June 4, so the OSS received no inflow during that period. Elevated water levels were measured in W9 on only a few occasions during this time. However, the water level in W9 decreased steadily during this period, demonstrating that OSS outflows were partially responsible for the water level in W9. Once OSS outflows resumed on June 4, the association between the water level in W9 and the tidal cycle became apparent once again, suggesting that some base level of septic tank outflow is required for the tidal cycle to consistently produce elevated water levels in W9. Ultimately, the interaction between septic tank effluent, infiltrating seawater and perhaps shallow groundwater determines the water depth in W9. The mixing dynamics among these waters could play an important role in N removal, because the OSS effluent is oxygenated by mixing with infiltrating seawater and then rapidly transported to Hood Canal as the tide retreats.

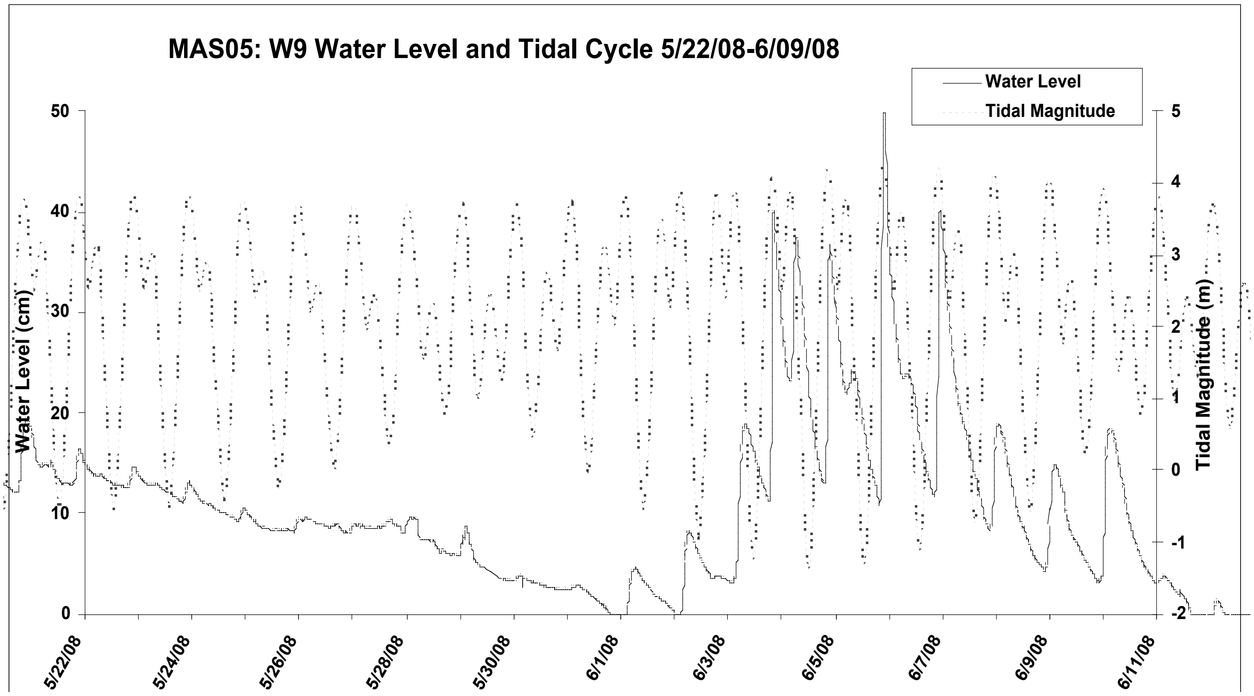


Figure 5. Water level in Well 9 (W9) at MAS05 from May 22 to June 11, 2008. The average standing water level in W9 was 10.1 cm. There were no OSS discharges from May 22 to June 4. Peak standing water levels typically corresponded to high tides. There were no OSS discharges from May 22 to June 4.

3.5.2.2 MAS07

Pressure transducer data were continuously collected from MAS07 Well 9 (W9), located 2.6 m up-gradient of the bulkhead at MAS07, between June 24 and July 23, 2008. The data series most relevant to experimental results, spanning June 29 to July 11, is displayed in Figure 6. Water was rarely detected in W9. The average water level was only 0.58 cm. However, substantial peaks appeared regularly in the data series, indicating that water was present to a depth of as much as 30 cm. The presence of water in W9 was very transient, as peaks rapidly appeared and subsided. Although elevated water levels corresponded to high tides, high tides did not necessarily result in increased water levels. Similar to MAS05, the water levels in W9 at MAS07 depends on the flows of both OSS effluent and infiltrating seawater, which both vary hourly and daily. Because there was no period during which the household was unoccupied, it was difficult to distinguish between OSS and seawater influences on the water levels in W9. Nevertheless, it is clear that seawater appears to regularly infiltrate the subsurface down-gradient of the OSS at MAS07. The degree of seawater infiltration at MAS07 is likely less than at MAS05, since the property at MAS07 is 1.4 m higher above the shoreline than is MAS05.

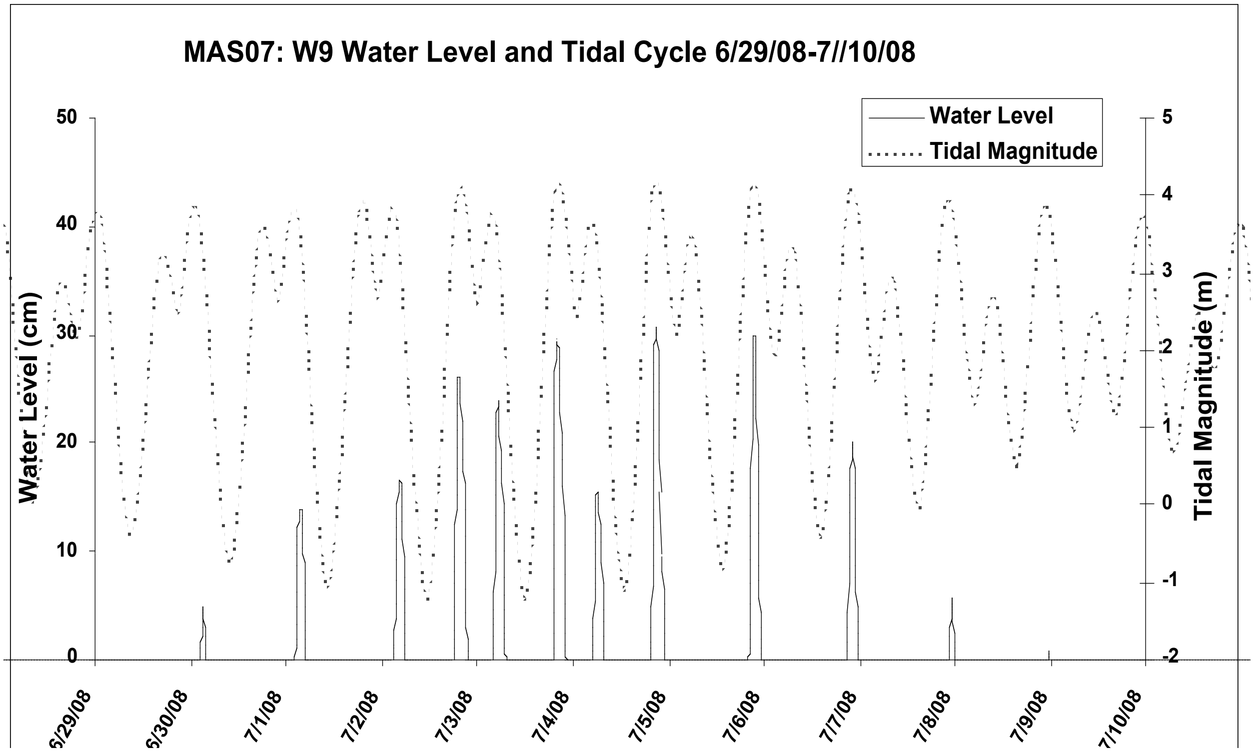


Figure 6. Water level measured in Well 9 at MAS07 from June 29 to July 11, 2008. The average standing water level in well 9 was 0.58 cm. Peak standing water depths were observed at the same time as high tides. However, high tides did not necessarily result in increased standing water levels.

3.5.3 Seawater Samples

The DIN and Br$^-$ concentrations measured in seawater and septic tank samples served as the endpoints for the mixing model. Sixteen seawater samples were collected and analyzed over the course of the three experiments. The data for these samples are presented in Table 3.

Table 3. Statistics pertaining to constituent concentrations measured in seawater samples. The DIN and Br$^-$ values in samples were used in the probabilistic mixing model.

Seawater Sample Concentrations											
Statistics	DIN (?g/L)	NO$_3$-N (?g/L)	NH$_4$-N (?g/L)	TDN (?g/L)	DON (?g/L)	Br$^-$ (mg/L)	I$^-$ (?g/L)	Li$^+$ (?g/L)	DOC (mg/L)	DO (mg/L)	Salinity (ppt)
Mean	70	31	38	222	162	51.4	63	145	1.4	10.7	27.3
STDEV	35	23	15	73	65	0.5	1.4	9	0.5	0.4	0.7
Median	56	19	35	191	133	51.4	63	148	1.6	10.8	27.5
n	12	12	12	16	12	16	10	16	16	12	12

3.5.4 MAS05 Experiment #1

3.5.4.1 Sample Collection

The first experiment at MAS05 was conducted April 9, 2008. Samples were collected hourly over an 11-hour period between high tides. The timing and magnitude of tides, as well as the duration of sampling, are displayed in Figure 7. The distribution of the 49 samples collected is shown in Table 4. Of the 42 well samples, 28 were collected prior to the 14:09 low tide. Wells 1, 2, 5 and 6, located up-gradient of the bulkhead (UGBH), yielded 18 samples, and Wells 3, 4 and 7, located in the intertidal zone down-gradient of the bulkhead (DGBH), produced 24 samples. The wells adjacent to the drainline (W8-W12) had yet to be installed at the time of this experiment. The sample collection success rate decreased as low tide approached and the saturated zone in the subsurface lowered. Samples were collected from all wells during the three sampling periods prior to 12:00 and again at 20:00. The samples series collected between these times were incomplete, as numerous wells failed to yield water. Samples could only be collected from W4 and W7 between 13:30 and 19:00.

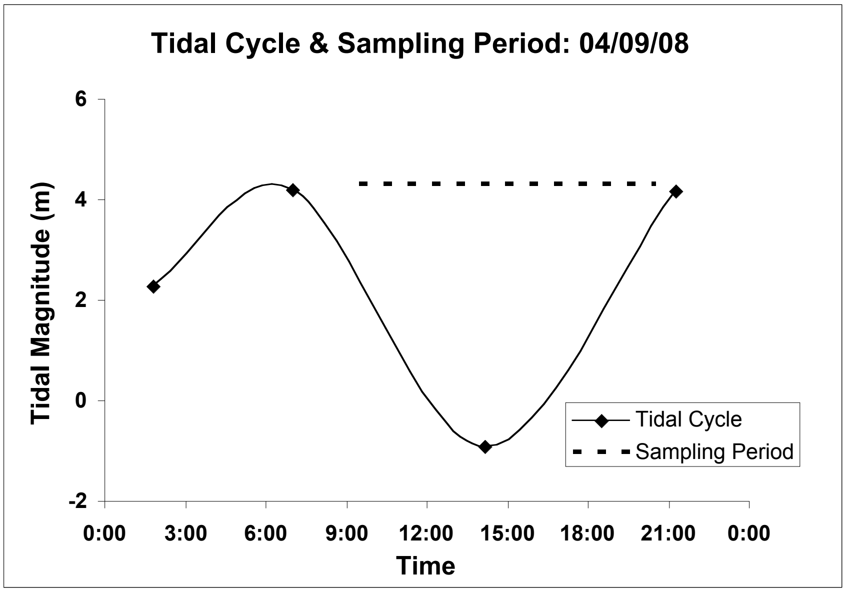


Figure 7. The tidal cycle and sample collection period pertaining to MAS05 experiment #1 conducted on April, 9 2008.

Table 4. Distribution of samples collected during MAS05 experiment #1. Wells 1, 2, 5 and 6 are UGBH. Wells 3, 4 and 7 are DGBH.

Sample Collection: 04/09/08	
Sample ID	**Samples Collected**
Well 1 (W1)	5
Well 2 (W2)	5
Well 3 (W3)	5
Well 4 (W4)	10
Well 5 (W5)	4
Well 6 (W6)	4
Well 7 (W7)	9
Seawater (SW)	4
Septic	3
Total	49

3.5.4.2 Results

The bromide concentrations of the well samples are displayed in Figure 8. The average Br^- concentration for wells adjacent to the bulkhead was 51.6±0.3 mg/L, and that in the three septic tank samples was 0.28±0.02 mg/L. The application of these Br^- values along with the average Br^- concentration measured in seawater to Equations 2 and 3 indicates that OSS effluent represented, on average, 0.057±0.143% of all sample volumes. The average OSS fractions of samples collected from wells UGBH and DGBH were 0.11%±0.19% and 0.02%±0.07%, respectively. These f_{SEP} values indicate that OSS effluent generally exerted a somewhat larger influence on wells UGBH than those DGBH.

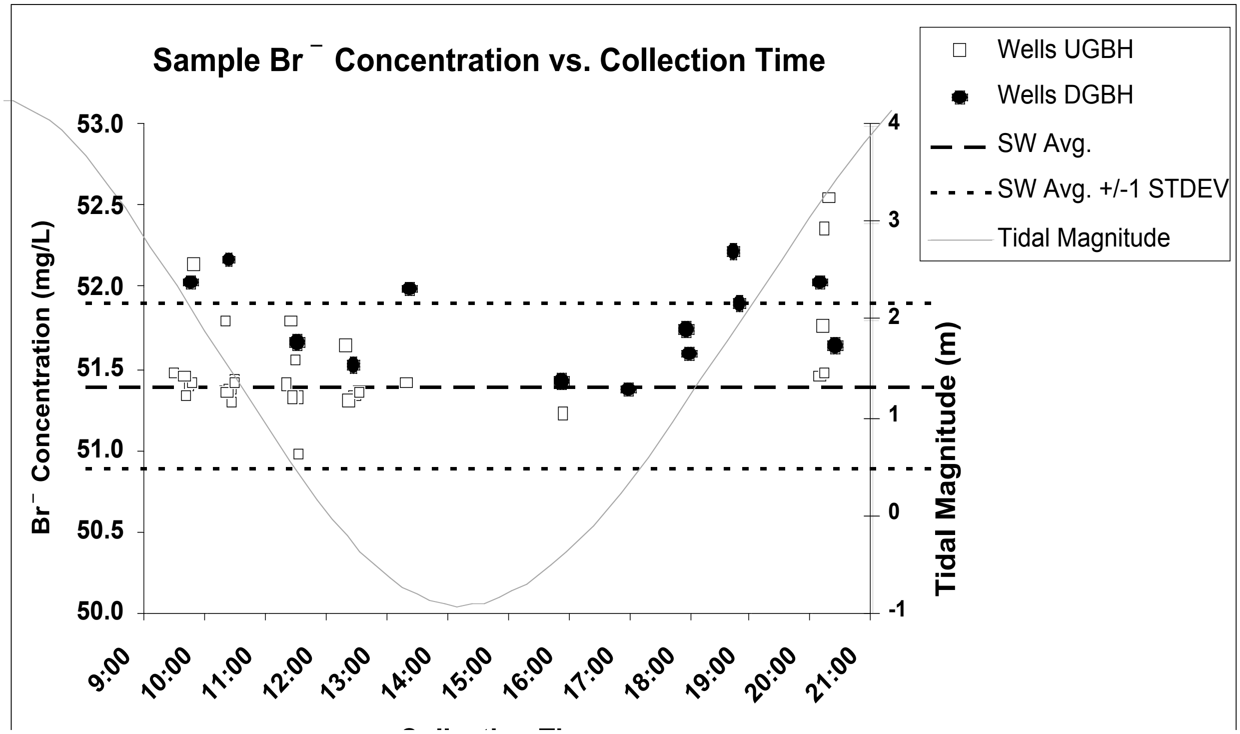


Figure 8. MAS05 experiment #1 Br^- concentrations plotted against sample collection time. The SW average and associated standard deviation pertain to all 16 seawater samples analyzed for Br^- over the course of the project.

Sample DIN concentrations are plotted in Figure 9. The mean DIN concentration in all well samples was 203±67 µg/L. Based on an average f_{SEP} value of 0.057%, the average DIN concentration associated with conservative transport of N was 138 µg/L. All DIN concentrations measured in well samples exceeded that value, indicating that N removal was probably minimal. The DIN data for wells were characterized by a high degree of scatter, however DIN concentrations in all samples were elevated considerably above the background seawater concentration of 70±35 µg/L. The average DIN concentration in the three septic tank samples was 120,000±1,460 µg/L.

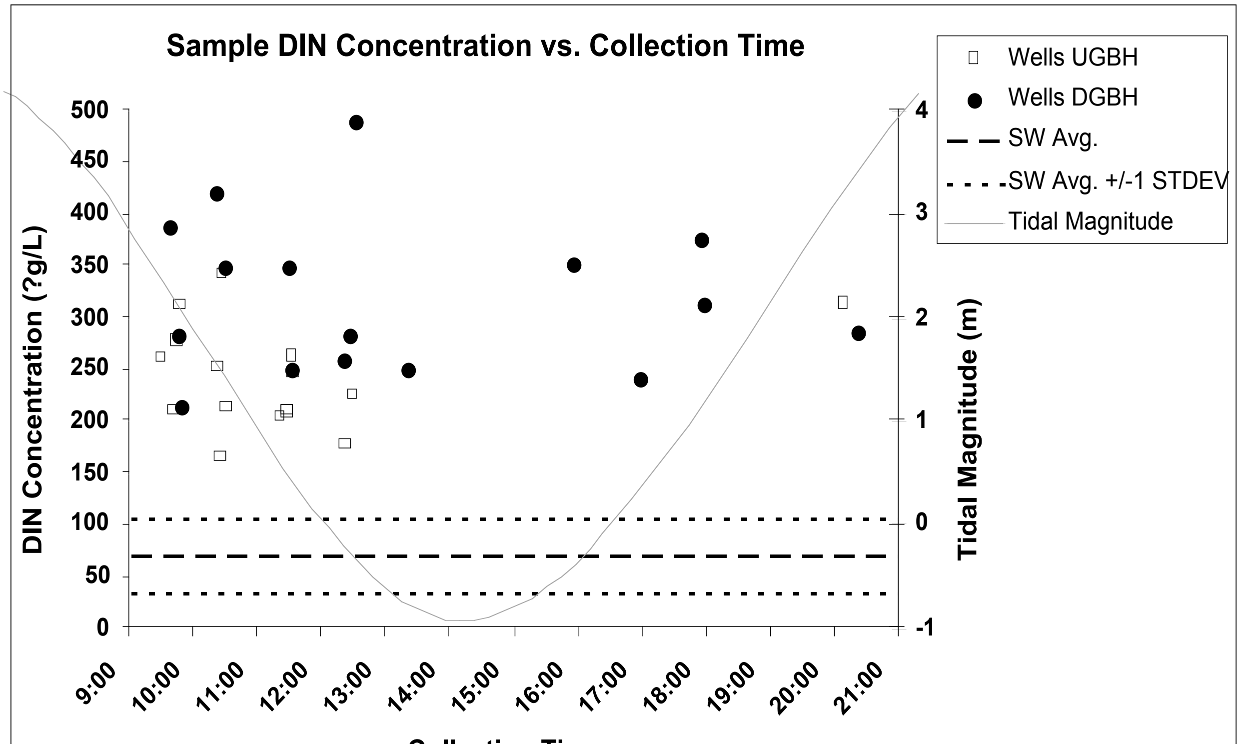


Figure 9. MAS05 experiment #1 DIN concentrations in samples plotted against sample collection time. The SW average and associated standard deviation pertain to all 12 seawater samples analyzed for DIN over the course of the project. The mean DIN concentration measured in all well samples was 203±67 µg/L. The expected average DIN concentration associated with conservative N transport was calculated as 138 µg/L based on the average f_{sep} value of 0.057%.

The average speciation of DIN in different sample points is displayed in order of increasing distance from the OSS in Figure 10. NH_4-N accounted for all the DIN in OSS effluent, but only 18±20% of the DIN in well samples, and 71±12% of the DIN in seawater, with the remainder accounted for by nitrate in all cases. Although the relative contribution of NH_4-N to DIN in well samples is smaller than that in seawater samples, the absolute NH_4-N concentrations in both sample sources are similar. The mean NH_4-N concentrations in well and seawater samples were 37±48 µg/L and 38±15 µg/L, respectively. The speciation of DIN in well samples from each series is shown in Figure 11, organized chronologically. The largest fractions of NH_4-N in well samples were in samples collected near the high tides at 7:00 and particularly at 21:18.

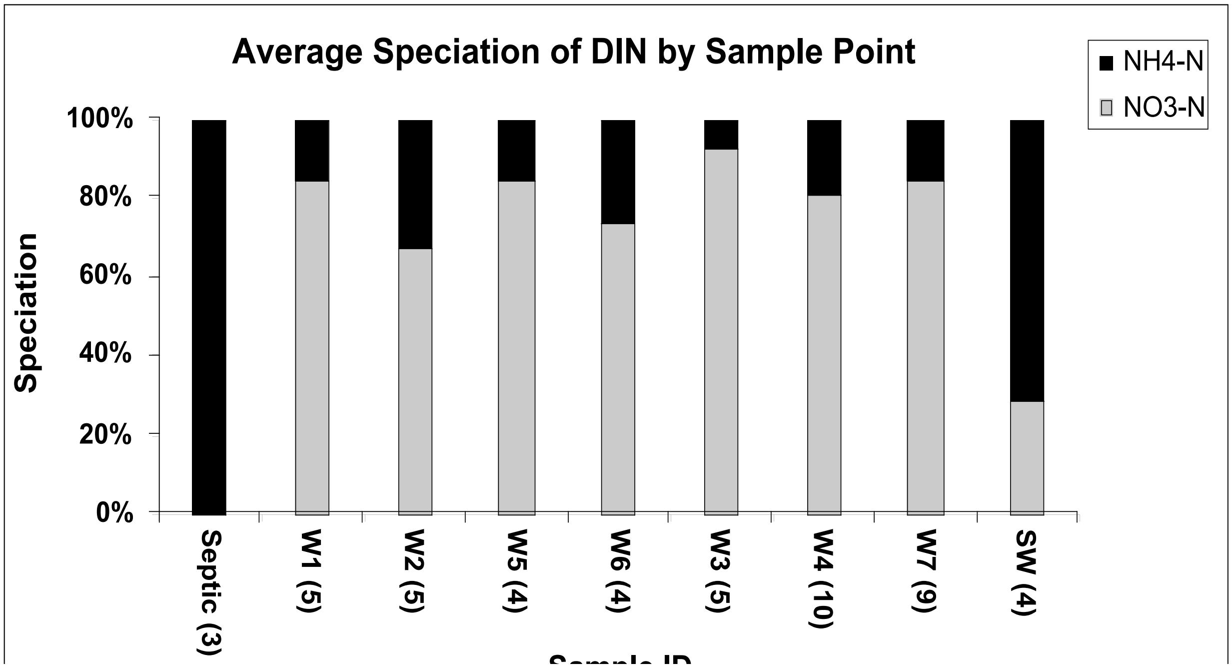


Figure 10. MAS05 experiment #1 average speciation of DIN by sample source, organized in order of distance from the OSS. The values in parentheses represent the number of samples collected from specific sampling points. The DIN in septic tank samples was entirely in the form of NH$_4$-N. The DIN in well samples was primarily composed of NO$_3$-N, accounting for 82% of DIN on average. Ammonium contributed 71±12% of DIN in seawater samples on average.

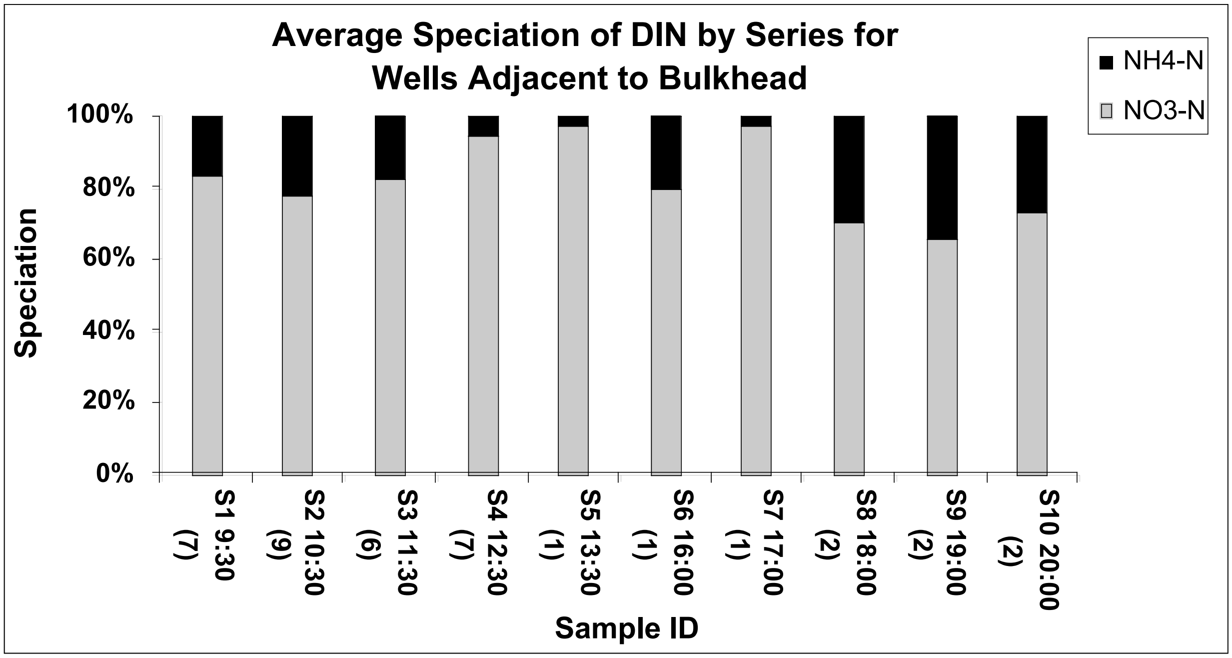


Figure 11. MAS05 experiment #1 average speciation of DIN in well samples collected during each series. Series are abbreviated as "S" and followed by the sample collection time. Values in parentheses represent the number of samples collected during particular series. The greatest fractions of NH$_4$-N in were associated with samples collected near the high tides at 7:00 and particularly at 21:18. Nitrate represented a greater portion of DIN in samples collected closer to the 14:09 low tide.

3.5.5 MAS05 Experiment #2

3.5.5.1 Sample Collection

The second experiment at MAS05 was conducted July 7, 2008. Samples were collected hourly over a 6.5-hour period beginning at the 8:14 high tide and ending at the 14:45 low tide. The timing and magnitude of tides, as well as the duration of sampling, are displayed in Figure 12. The distribution of the 67 samples collected is shown in Table 5. Of the 56 well samples collected, 21 were from the upper wells located behind the bulkhead (W1, W2, W5, W6); and 35 were from the lower wells in the intertidal zone (W3, W4, W7). The sample collection success rate decreased markedly as low tide approached. Samples were collected from all wells during the five series prior to and at 11:30. The sampling series conducted afterwards were incomplete, as numerous wells failed to yield water. Of the seven wells adjacent to the bulkhead, samples could only be collected from W4 and W7 from 13:30 onward.

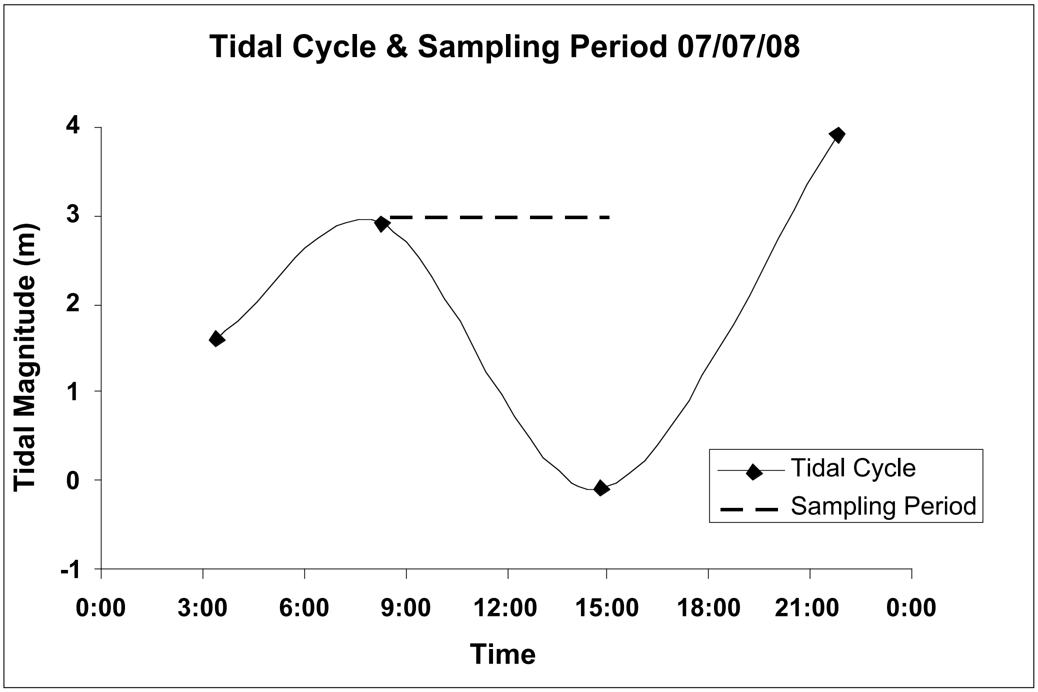


Figure 12. The tidal cycle and sample collection period pertaining to MAS05 experiment #2 conducted on July, 7 2007. The period of sample collection lasted 6.5 hours.

Table 5. The distribution of samples collected during MAS05 experiment #2. Wells 1, 2, 5 and 6 are UGBH. Wells 3, 4 and 7 are DGBH. Wells 8 and 10 are adjacent to the drainline.

Sample Collection: 07/07/08	
Sample ID	**Samples Collected**
Well 1 (W1)	5
Well 2 (W2)	6
Well 3 (W3)	6
Well 4 (W4)	8
Well 5 (W5)	5
Well 6 (W6)	5
Well 7 (W7)	7
Well 8 (W8)	8
Well 10 (W10)	6
Seawater (SW)	6
Septic	5
Total	67

3.5.5.2 Results

The samples collected during the second experiment at MAS05 were analyzed for iodide (I^-) in addition to bromide. Iodide was used as a secondary tracer to confirm results based on Br^- concentrations. Both Br^- and I^- concentrations indicate that the samples from wells near the drainline (W8 and W10) were 100% OSS effluent. However, the influence of the OSS was barely discernible in samples collected from wells adjacent to the bulkhead (W1 through W7). The Br^- concentrations near the bulkhead and near the drainline are displayed in Figures 13 and 14, respectively. The Br^- concentrations adjacent to the bulkhead decreased with time as the tide retreated. The linear regression R^2 values characterizing these trends were 0.48 for wells UGBH and 0.60 for wells DGBH. The I^- concentrations adjacent to the bulkhead also decreased with time, with R^2 values of 0.47 and 0.53 for wells UGBH and DGBH, respectively (Figures 15 and 16). When analyzed using Equations 2 and 3, the decreases in Br^- and I^- concentrations over time yielded decreasing f_{SW} values and increasing f_{SEP} values. These results indicate that, as the tide retreated, the influence of the OSS effluent on subsurface water increased.

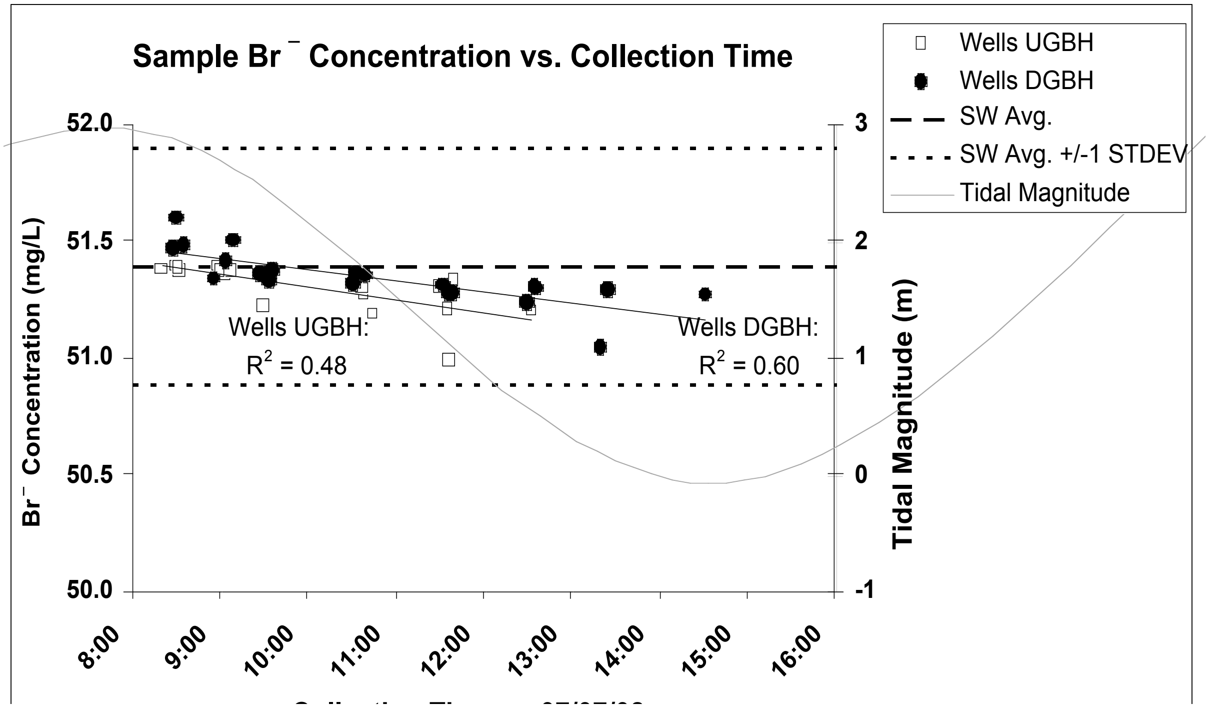


Figure 13. MAS05 experiment #2 Br$^-$ concentrations in samples collected from wells adjacent to the bulkhead plotted against sample collection time. The SW average and associated standard deviation pertain to all 16 seawater samples analyzed for Br$^-$ over the course of the project. The Br$^-$ concentrations measured in wells adjacent to the bulkhead decreased with time, indicating that the influence of the OSS effluent on subsurface water increased as the tide retreated.

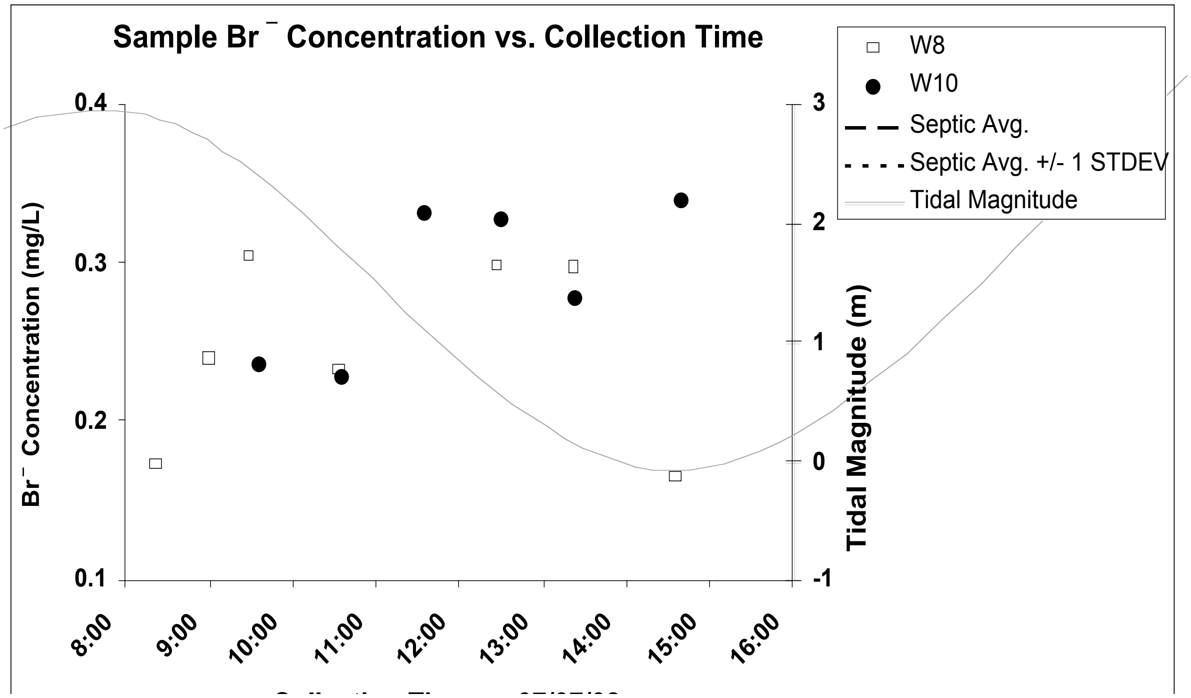


Figure 14. MAS05 experiment #2 Br$^-$ concentrations in samples collected from wells adjacent to the drainline plotted against sample collection time. The septic tank average and associated standard deviation only

pertain to the 5 samples collected during the experiment. Note that scale for the y-axis in Figure 14 is different from that in Figure 13.

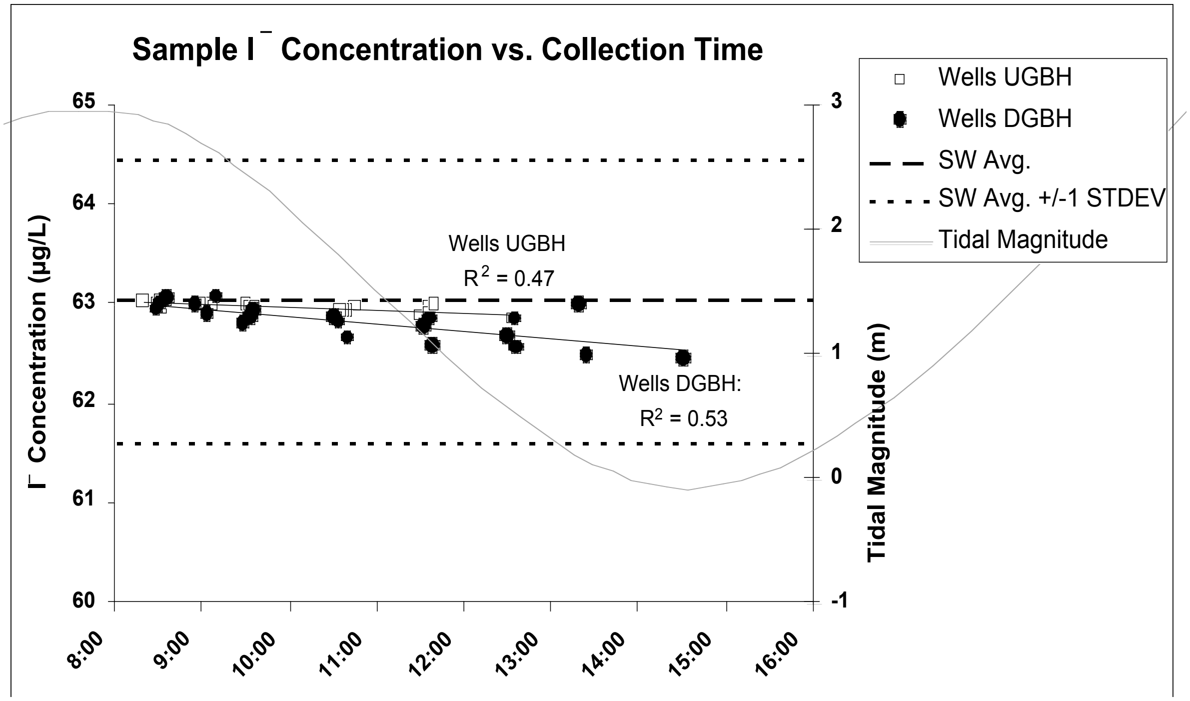


Figure 15. MAS05 experiment #2 I¯ concentrations in samples collected from wells adjacent to the bulkhead plotted against sample collection time. The SW average and associated standard deviation pertain to all 10 seawater samples analyzed for I¯ over the course of the project. Similar to Br¯ concentrations, the I¯ concentrations measured in wells adjacent to the bulkhead decreased with time, indicating that the influence of the OSS effluent on subsurface water increased as the tide retreated.

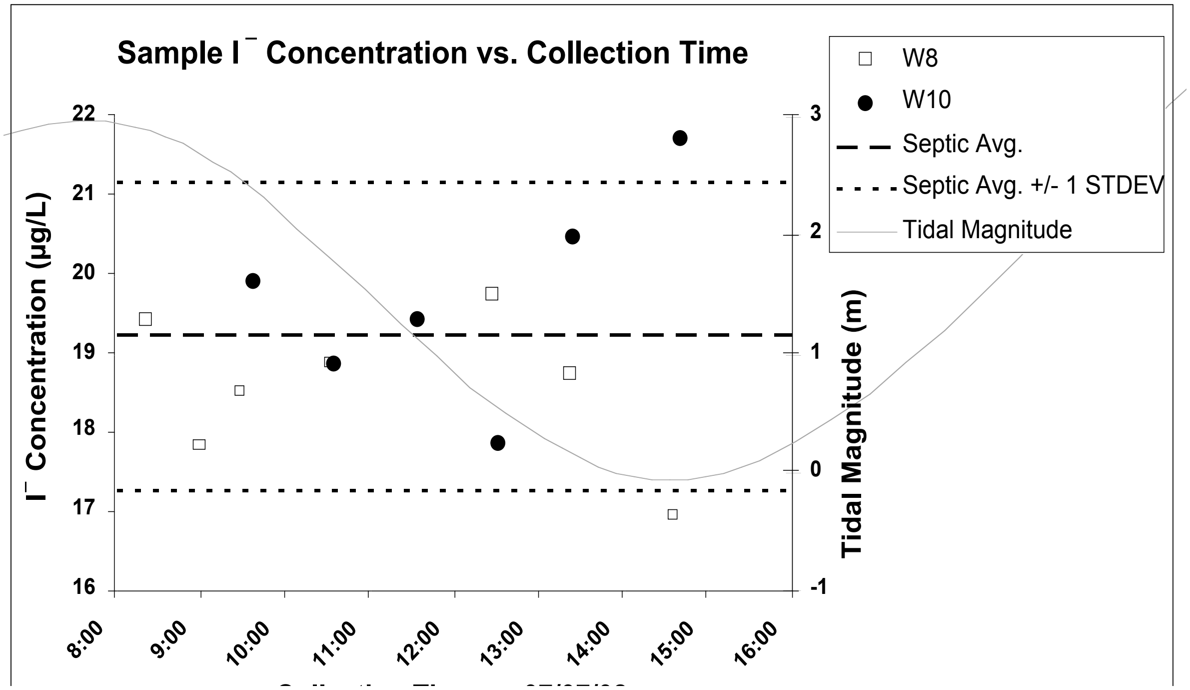


Figure 16. MAS05 experiment #2 I¯ concentrations in samples collected from wells adjacent to the drainline plotted against sample collection time. The septic tank average and associated standard deviation only

pertain to the 5 samples collected during the experiment. Note that scale for the y-axis in Figure 16 is different from that in Figure 15.

The mean Br^- and I^- concentrations in the five septic tank samples were 0.29±0.03 mg/L and 19.2±1.9 µg/L, respectively. Based on Equations 2 and 3, OSS effluent comprised the entire sample volume collected from wells adjacent to the drainline. The estimated OSS fractions in different groups of wells are presented in Table 6. Iodide-based f_{SEP} values were computed by substituting the appropriate I^- concentrations into Equation 2 and subsequently solving Equation 3. Some discrepancies exist between the OSS fractions computed using the different tracers, but the values computed using both approaches are comparable. Ultimately, the Br^-- and I^--based OSS fractions were combined to yield an average f_{sep} of 0.23±0.16% for samples from wells adjacent to the bulkhead. The mean OSS fractions for samples UGBH and DGBH were 0.28±0.18% and 0.17%±0.13, respectively. The overall average f_{sep} value was nearly four times the value of 0.057±0.143% computed using data from the first experiment. This difference might be due to the fact that, although both sampling periods started near high tide, the first experiment continued to the next high tide, while the second experiment ended at low tide. The influence of infiltrating seawater from the returning high tide decreased f_{sep} values for samples collected during that time period in the first experiment and consequently decreased the average f_{sep} value. However, the f_{sep} values pertaining to samples collected at similar times relative to the tidal cycle during both experiments followed the same pattern. During both experiments, the f_{sep} values were lowest in samples collected during the first three series nearest to high tide (0.077±0.074% and 0.098±0.082%, respectively) and they gradually increased up to the point of low tide.

Table 6. OSS fraction (f_{sep}) computed using Br^- data and I^- data. The Br^- and I^- OSS fractions were averaged to yield a mean OSS fraction value.

Well Group	**Br^--Based f_{sep}**		**I^--Based f_{sep}**		**Br^- & I^- Average f_{sep}**	
	% Septic Avg	**STDEV**	**% Septic Avg**	**STDEV**	**% Septic Avg**	**STDEV**
Near OSS	100	0.1	100	2.2	100	0.4
Near Bulkhead	0.157	0.174	0.291	0.247	0.225	0.160
UGBH	0.143	0.157	0.413	0.286	0.281	0.174
DGBH	0.170	0.193	0.169	0.109	0.169	0.126

Sample DIN concentrations for wells adjacent to the bulkhead are plotted in Figures 17 and 18. The average DIN concentrations in samples from wells UGBH and DGBH were 313±49 µg/L and 506±140 µg/L, respectively. The mean DIN concentration for all wells adjacent to the bulkhead was 409±142 µg/L. Based on an average f_{sep} value of 0.23%, the average DIN concentration expected for conservative N transport was 313 µg/L. The DIN concentrations in many well samples exceeded that value, suggesting that N removal was probably low. DIN concentrations in all well samples were significantly above the background seawater concentration of 70±35 µg/L. As shown in Figure 18, the DIN concentrations in six of the seven wells adjacent to the bulkhead increased with time, with R^2 values ranging from 0.67 to 0.93. Like the Br^- and I^- data, the DIN concentrations indicate that, as the tide retreated, the contribution of the OSS effluent to subsurface water increased. Sample DIN concentrations for wells adjacent to the drainline are plotted in Figure 19. The mean DIN concentration in samples from wells adjacent to the drainline was 97,300±8280 µg/L. The average DIN concentration in the three septic tank samples was 108,000±4230 µg/L.

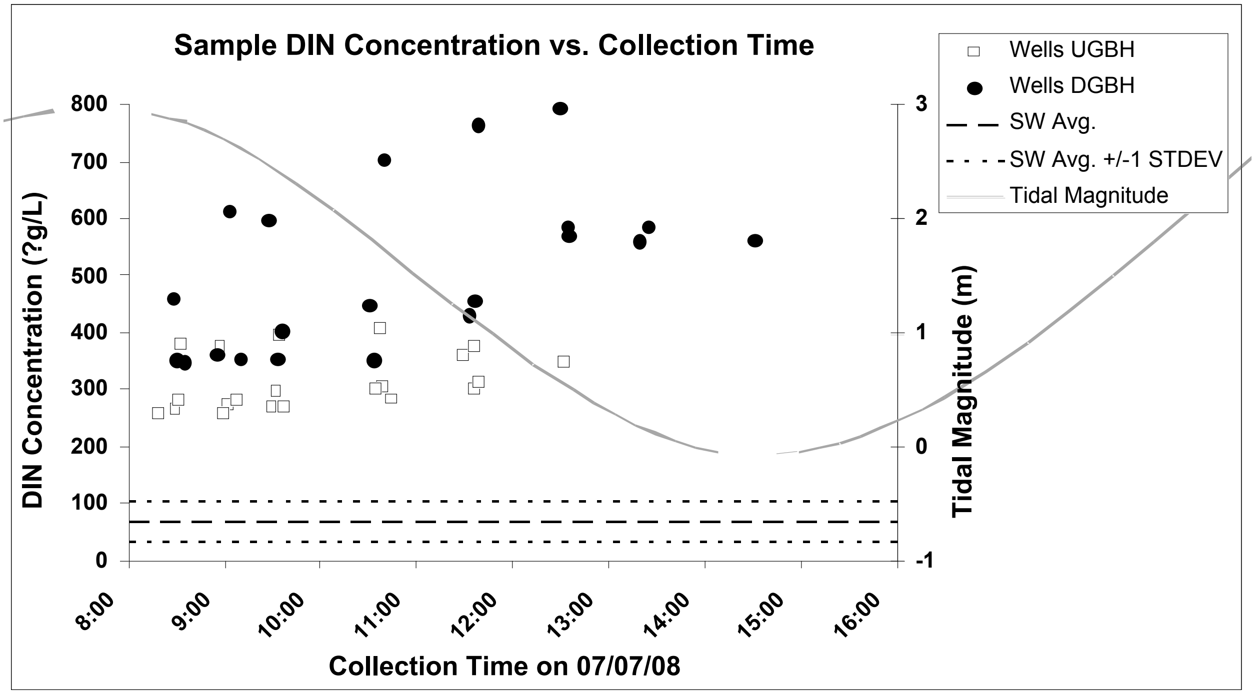


Figure 17. MAS05 experiment #2 DIN concentrations measured in samples collected from wells adjacent to the bulkhead plotted against sample collection time. The SW average and associated standard deviation pertain to all 12 seawater samples analyzed for DIN over the course of the project. The mean DIN concentration measured in all samples from wells UGBH & DGBH was 409±142 µg/L. The expected average DIN concentration associated with conservative N transport was calculated as 313 µg/L based on the average f_{sep} value of 0.023%.

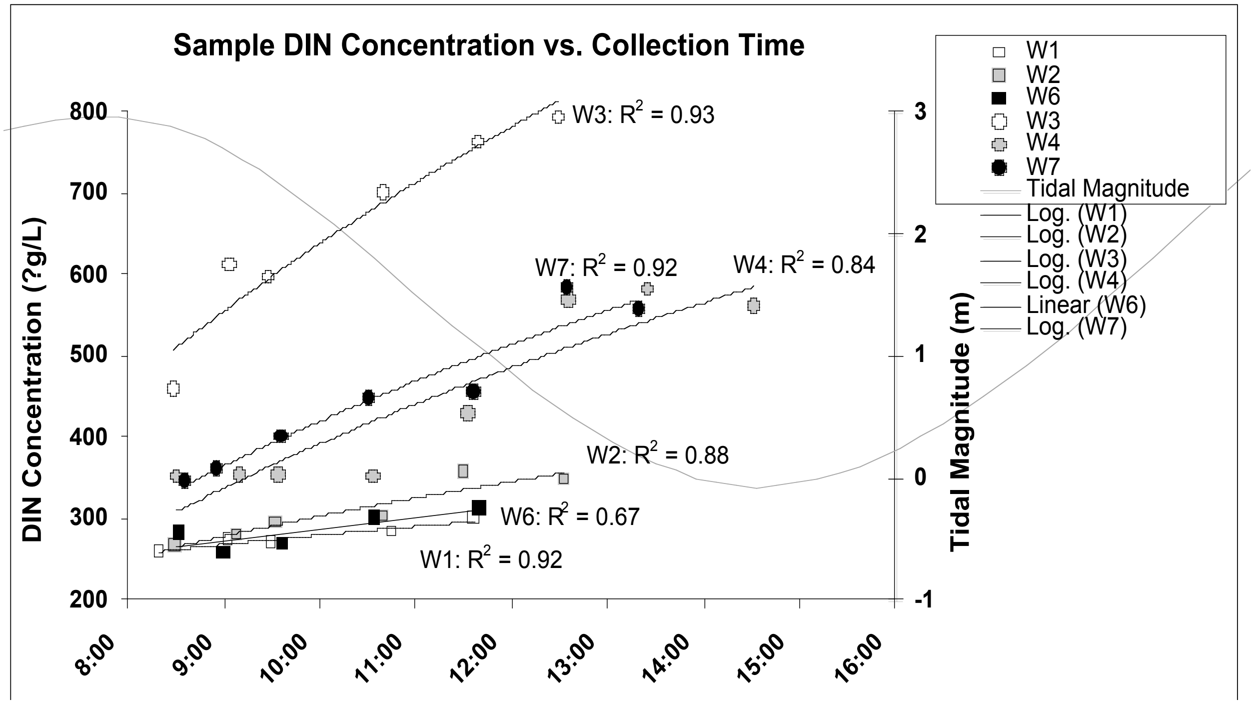


Figure 18. MAS05 experiment #2 trends in DIN concentrations versus collection time for individual wells. The DIN concentrations in six of the seven wells adjacent to the bulkhead increased with time, indicating that the influence of the OSS effluent on subsurface water increased as the tide retreated. Squares and circles are used to symbolize wells UGBH and DGBH, respectively.

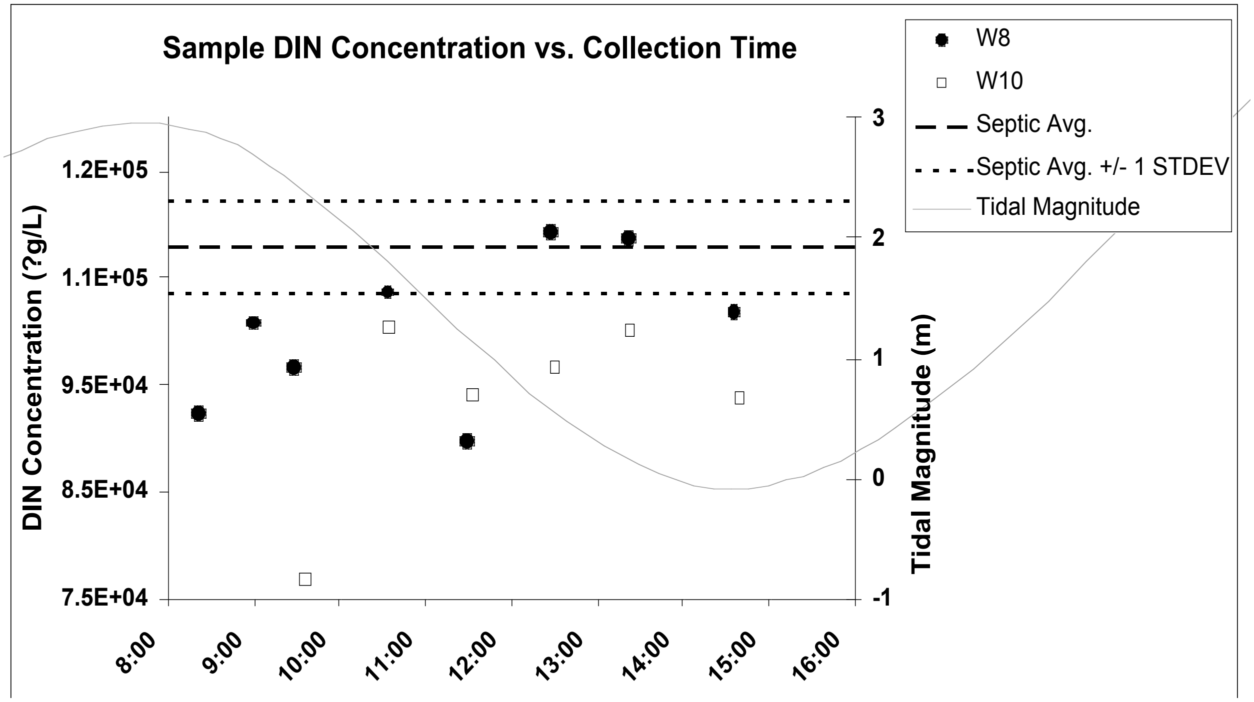


Figure 19. MAS05 experiment #2 DIN concentrations adjacent to the drainline plotted against sample collection time. The septic tank average and associated standard deviation only pertain to the 5 samples collected during the experiment. Note that the scale on the y-axis in Figure 19 differs from that in Figures 17 and 18.

The average speciation of DIN in samples from different locations is shown in order of increasing distance from the OSS in Figure 20. The DIN in septic tank effluent and in samples from wells adjacent to the drainline (W8 and W10) was entirely NH_4-N, indicating that nitrification did not occur between the drainline and wells W8 and W10. Similar to the results of MAS05 experiment #1, NO_3-N represented the predominant form of DIN in samples from wells adjacent to the bulkhead. However, NO_3-N was even more prevalent in those samples during the second experiment, accounting for 96±3.2% of the DIN. Although the relative contribution of NH_4-N to DIN in well samples is smaller than that in seawater samples, the absolute NH_4-N concentrations in both sample sources are similar. The mean NH_4-N concentrations in well and seawater samples were 32±44 µg/L and 38±15 µg/L, respectively. The DIN in seawater samples was almost evenly split between NH_4-N and NO_3-N, with NH_4-N accounting for 54±16% of the DIN, on average. This represented a departure from the DIN speciation observed in seawater samples during the first experiment at MAS05, in which NH_4-N contributed 71±12% of the DIN. The average speciation of DIN in samples from wells adjacent to the bulkhead is chronologically organized by series in Figure 21. Nitrate represented the overwhelming majority of DIN in samples from wells adjacent to the bulkhead, and no temporal pattern in DIN speciation was evident.

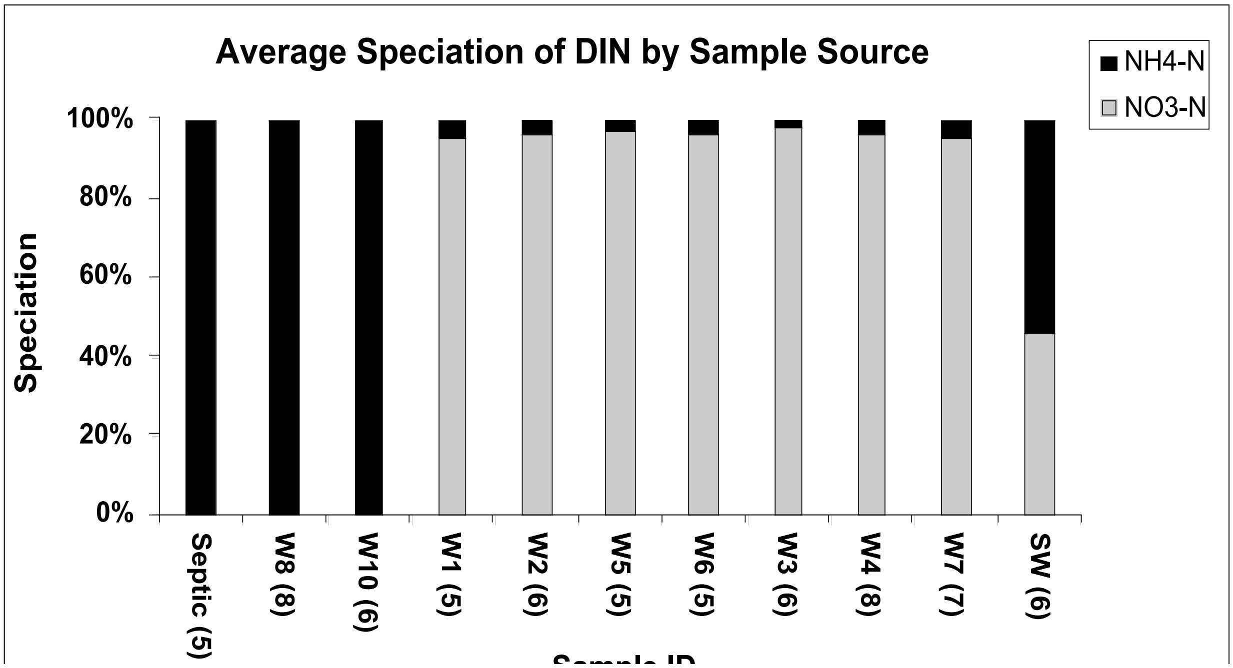


Figure 20. MAS05 experiment #2 average speciation of DIN by sample source organized in order of increasing distance from the OSS. The values in parentheses represent the number of samples collected from specific sampling sources. The DIN in septic tank samples and samples from wells adjacent to the drainline was entirely in the form of NH_4-N. The DIN in samples from wells adjacent to the bulkhead was primarily composed of NO_3-N which accounted for 96% of DIN on average. The DIN in seawater samples was split slightly in favor of NH_4-N on average, but the NH_4:NO_3 ratio ranged from 1 to 4 in individual samples.

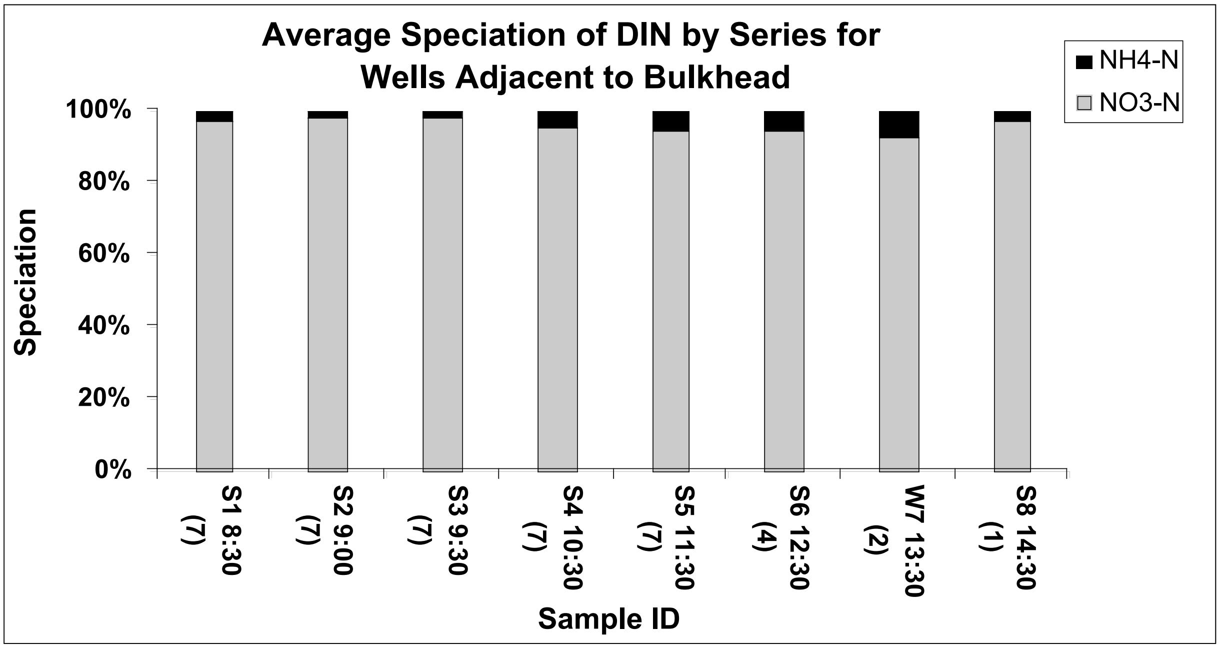


Figure 21. MAS05 experiment #2 average speciation of DIN in samples from wells adjacent to the bulkhead collected during each series. Series are abbreviated as "S" and followed by the sample collection time. The values in parentheses represent the number of samples collected during particular series.

3.5.6 MAS07 Experiment #3

3.5.6.1 Sample Collection

A third experiment was conducted at MAS07 on the night of June 23, 2008. Samples were collected every half-hour during a 5.5-hour window surrounding the 3.8-m high tide at 21:52. The timing and magnitude of tides, as well as the duration of sampling, are displayed in Figure 22. The distribution of the 23 samples collected is provided in Table 7. Of the 14 well samples collected, only four were collected prior to the 21:52 low tide. These four samples were from wells adjacent to the OSS drainline (W1 and W2). All nine samples from wells adjacent to the bulkhead (W9, W10, W12) were collected during the two hours immediately following the high tide. The other four wells located within 2.6 m of the bulkhead never yielded water. The water level in W9, W10 and W12 decreased as the tide retreated, and all wells were dry by 00:30 on June 24, 2008.

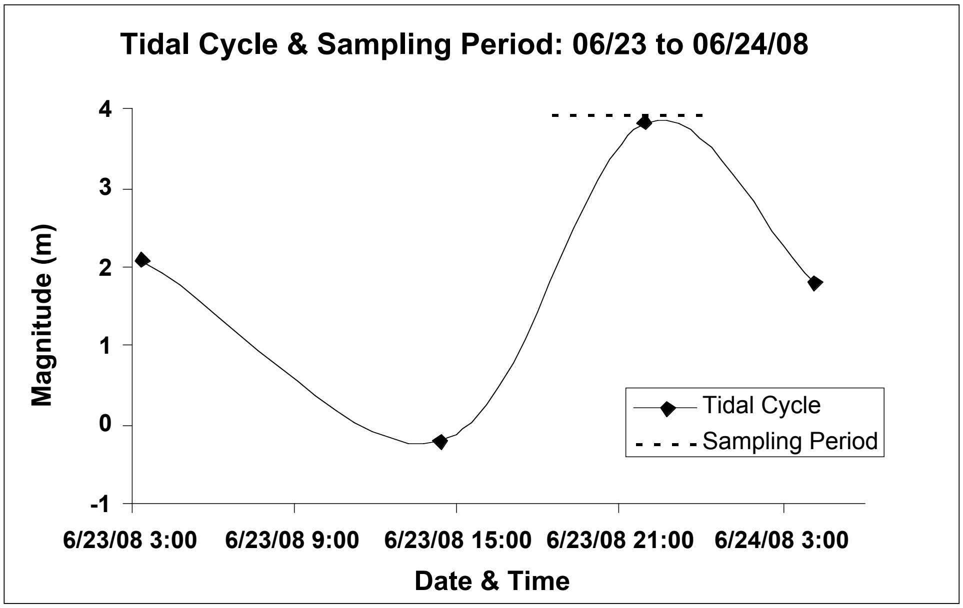


Figure 22. The tidal cycle and sample collection period pertaining to MAS07 experiment #3 conducted on the night of June 23, 2008. The sampling period lasted 5.5 hours.

Table 7. The distribution of samples collected during MAS07 experiment #3. Wells 1 and 2 are adjacent to the drainline. Wells 9, 10 and 12 are adjacent to the bulkhead.

Sample Collection: 06/23/08	
Sample ID	**Samples Collected**
Well 1 (W1)	3
Well 2 (W2)	2
Well 9 (W9)	3
Well 10 (W10)	4
Well 12 (W12)	2
Seawater (SW)	6
Septic	3
Total	23

3.5.6.2 Results

The results demonstrate that the OSS had a very strong influence on wells near the drainline (W1 and W2). However, the OSS signal was greatly diminished in samples collected from wells adjacent to the bulkhead (W9, W10 and W12) due to dilution by infiltrating seawater. Bromide concentrations in samples collected near the drainline and near the bulkhead are displayed in Figures 23 and 24, respectively. The mean Br^- concentration in three septic tank samples was 0.185±0.033 mg/L. Based on Equations 2 and 3, OSS effluent represented 100% of the samples from W1 and W2. The mean OSS fractions of samples from W9, W10 and W12 were 0.79±0.23%, 0.28±0.35% and 0.24±0.09%, respectively. The average f_{SEP} value for these three wells adjacent to the bulkhead was 0.39±0.35%.

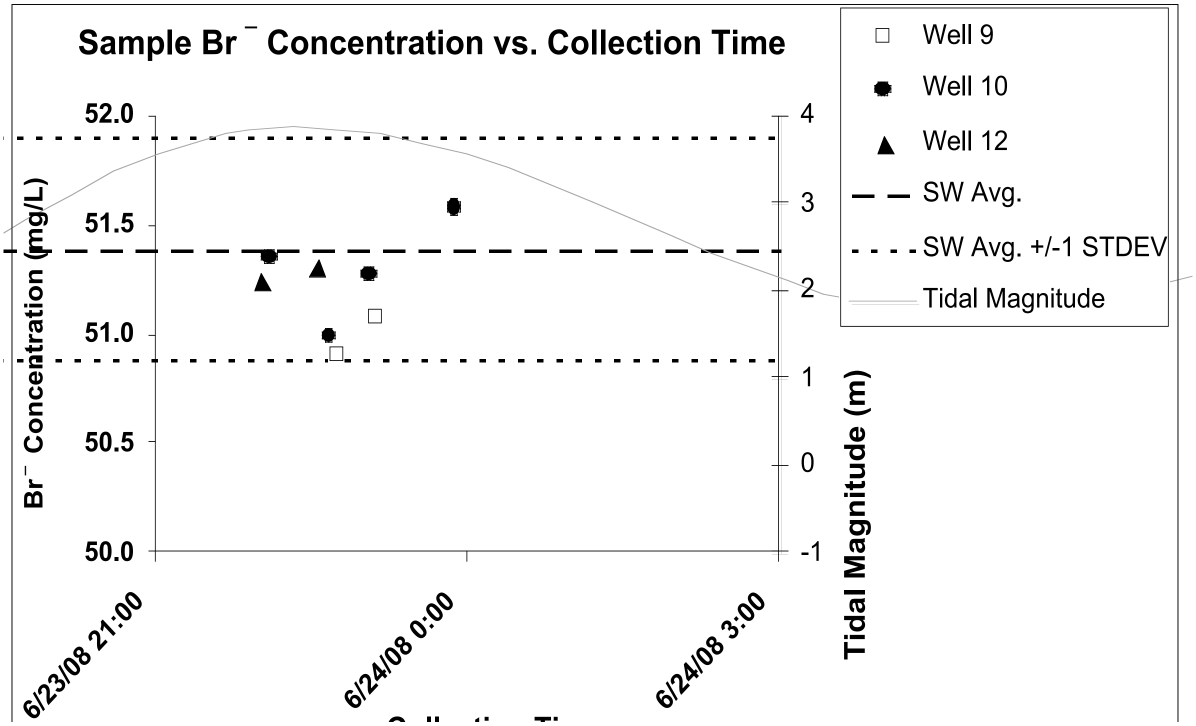


Figure 23. MAS07 experiment #3 Br⁻ concentrations measured in samples collected from wells adjacent to the bulkhead plotted against sample collection time. The SW average and associated standard deviation pertain to all 12 seawater samples analyzed for DIN over the course of the project.

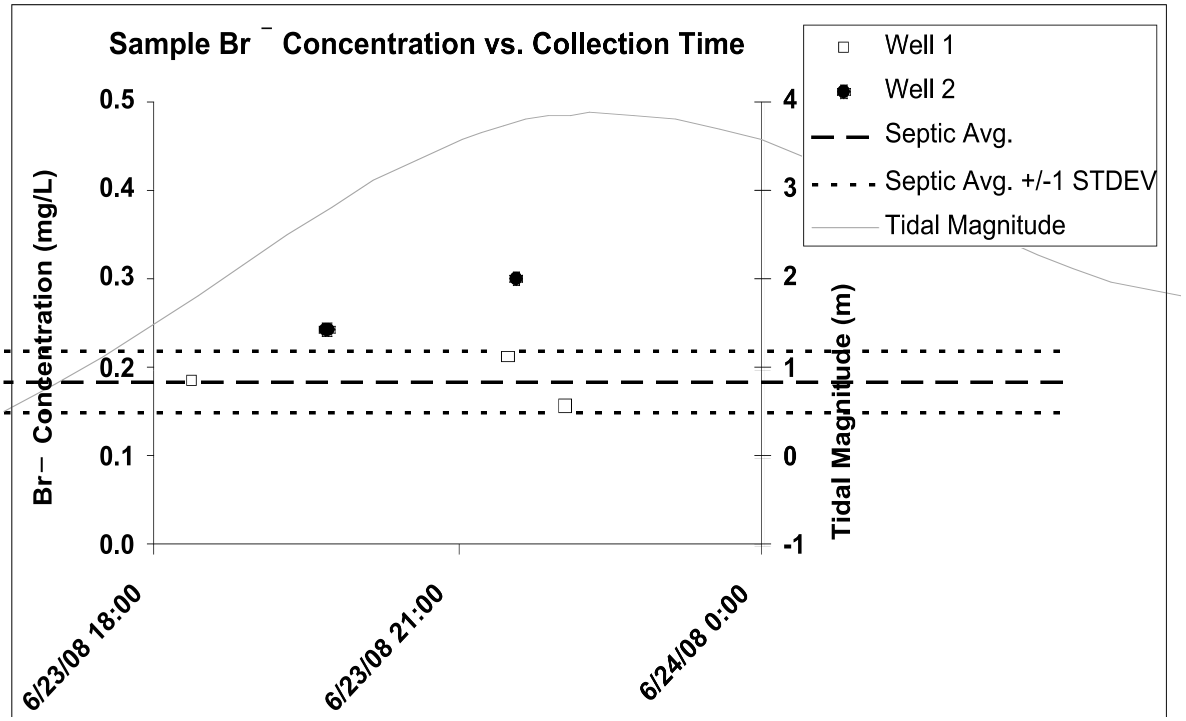


Figure 24. MAS07 experiment #3 Br⁻ concentrations measured in samples collected from wells adjacent to the bulkhead plotted against sample collection time. The septic tank average and associated standard deviation only pertain to the 3 samples collected during the experiment. Note that scale on the y-axis in Figure 24 is different from that in Figure 23.

Sample DIN concentrations are displayed in Figures 25 and 26. The mean DIN concentrations in samples from wells adjacent to the drainline and wells UGBH were 59,900±4,330 µg/L and 357±206 µg/L, respectively. Based on the average f_{sep} value of 0.394% for wells UGBH, the average DIN concentration associated with conservative transport of N was 259 µg/L. The DIN concentrations of numerous samples from wells adjacent to the bulkhead were near or greater than that value, indicating that N removal was minimal. Among the wells adjacent to the bulkhead, the DIN concentrations in all samples from W9 and W12 were well above background seawater levels. The average DIN concentration in the three OSS samples was 48,100±4780 µg/L.

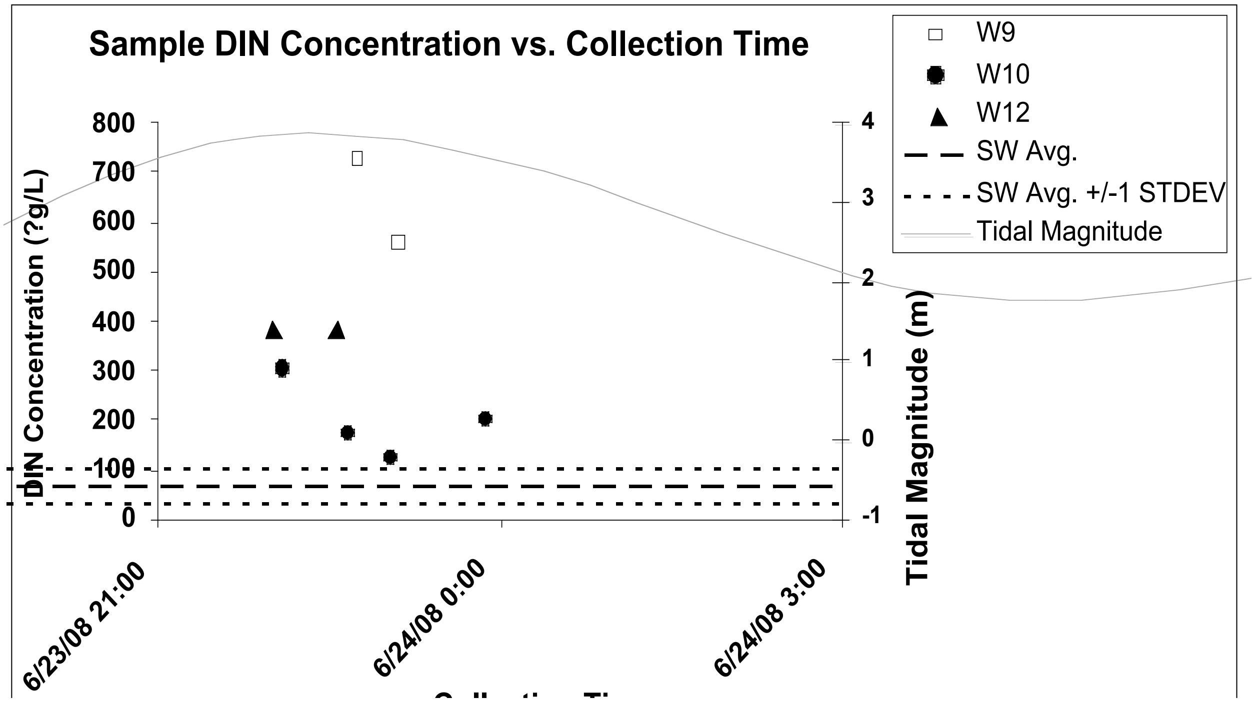


Figure 25. MAS07 experiment #3 DIN concentrations adjacent to the bulkhead plotted against sample collection time. The SW average and associated standard deviation pertain to all 12 seawater samples analyzed for DIN over the course of the project. The mean DIN concentration measured in these well samples was 357±206 µg/L. The expected average DIN concentration associated with conservative N transport was calculated as 259 µg/L based on an average f_{sep} value of 0.394%.

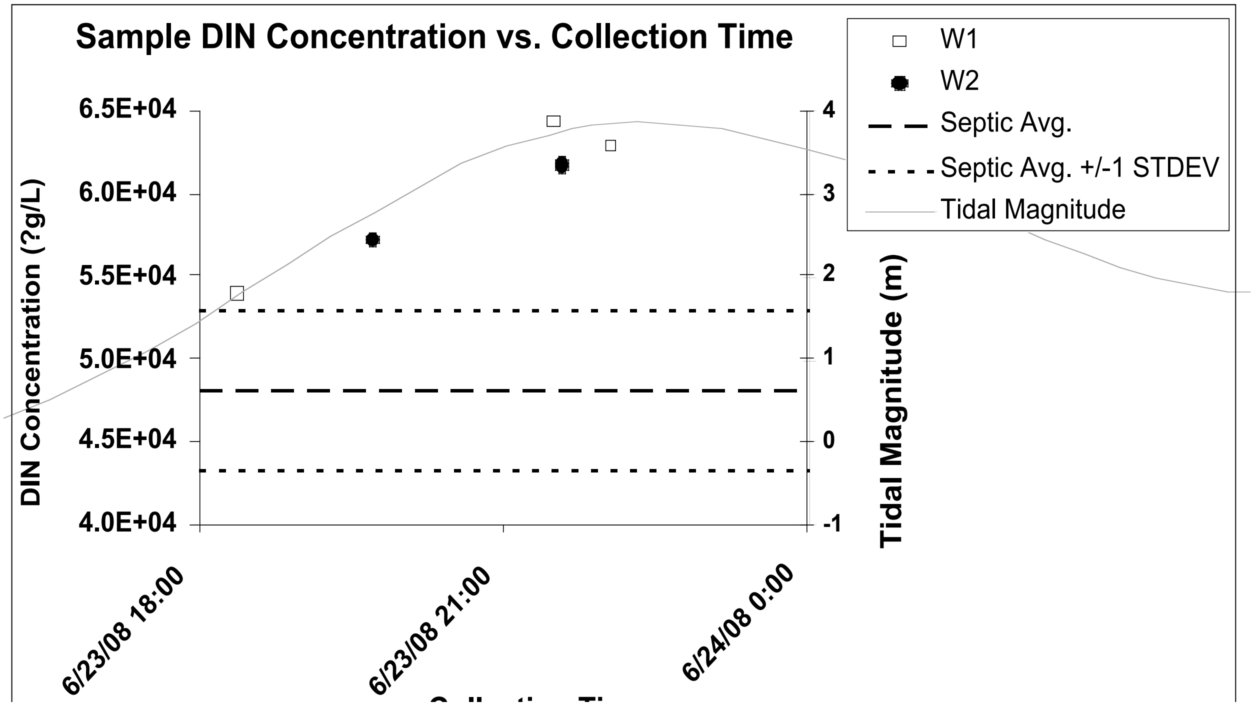


Figure 26. MAS07 experiment #3 DIN concentrations adjacent to the drainline plotted against sample collection time. The septic tank average and associated standard deviation only pertain to the 3 samples collected during the experiment. Note that the scale on the y-axis in Figure 26 is different from that in Figure 25.

The average speciation of DIN in different sample sources is displayed in order of increasing distance from the OSS in Figure 27. The DIN in samples from the septic tank and the well closest to the drainline (W1) was entirely NH_4-N. Approximately 6% of the DIN in W2 was NO_3-N. These results indicate that the nitrification of OSS effluent began at some point between Wells 1 and 2. Nitrate represented the predominant form of DIN in samples from wells adjacent to the bulkhead, accounting for 97±4.2% of the DIN on average. Ammonium was the predominant form of DIN in seawater samples. On average, NH_4-N comprised 62±6.0% of DIN in seawater samples. Despite that the relative contribution of NH_4-N to DIN in well samples is smaller than that in seawater samples, the absolute NH_4-N concentrations in those sample sources were 29±22 µg/L and 38±15 µg/L, respectively.

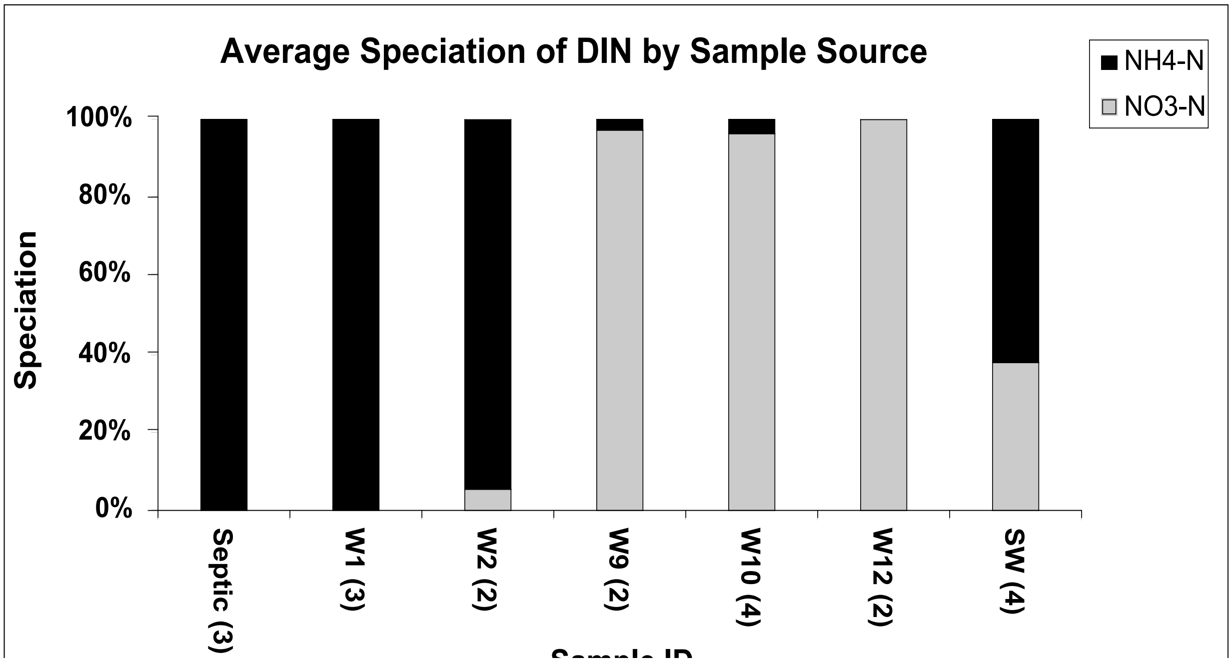


Figure 27. MAS07 experiment #3 average speciation of DIN by sample source organized in order of increasing distance from the OSS. The values in parentheses represent the number of samples collected from specific sampling points. The DIN in samples from the septic tank and W1 was entirely in the form of NH_4-N. A small fraction of DIN contained in OSS effluent was nitrified prior to reaching W2. The DIN in samples from wells adjacent to the bulkhead was primarily composed of NO_3-N which accounted for 97% of DIN on average. NH_4-N comprised 62±6.0% of the DIN in seawater samples.

3.6. Shoreline Sites Data Analysis

3.6.1 MAS05 Experiment #1

A comprehensive assessment of DIN removal probabilities for all samples collected from wells adjacent to the bulkhead is displayed in Figure 28. Conservative transport, characterized by DIN removal of less than or equal to 0%, represented the most probable fate of N released from the OSS with a likelihood of 65%. Nevertheless, the analysis suggested that the probability of >70% removal was nearly 25%. The median removal percentages calculated for chronologically collected samples did not correlate with the tidal sequence (ref: Figure 29), so the tidal sequence did not appear to be directly responsible for the variability in N removal.

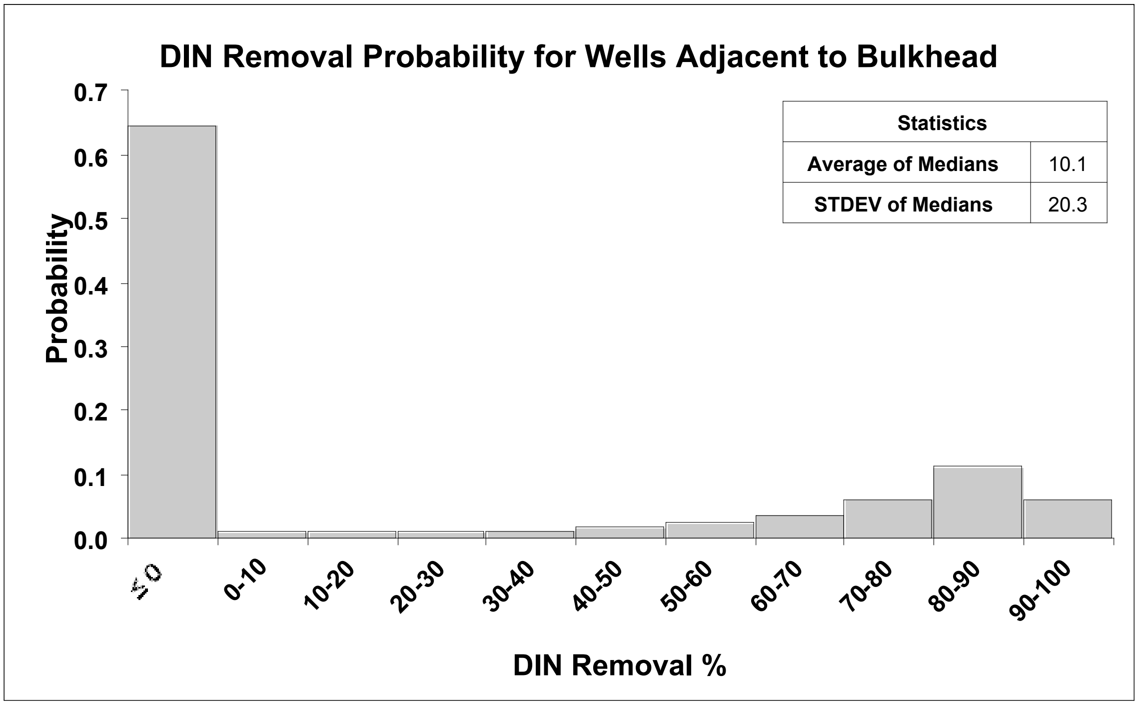


Figure 28. MAS05 experiment #1 DIN removal histogram for wells adjacent to the bulkhead. The data are skewed towards conservative transport of N. The probability that the N in OSS effluent was conservatively transported to the wells adjacent to the bulkhead was 65%.

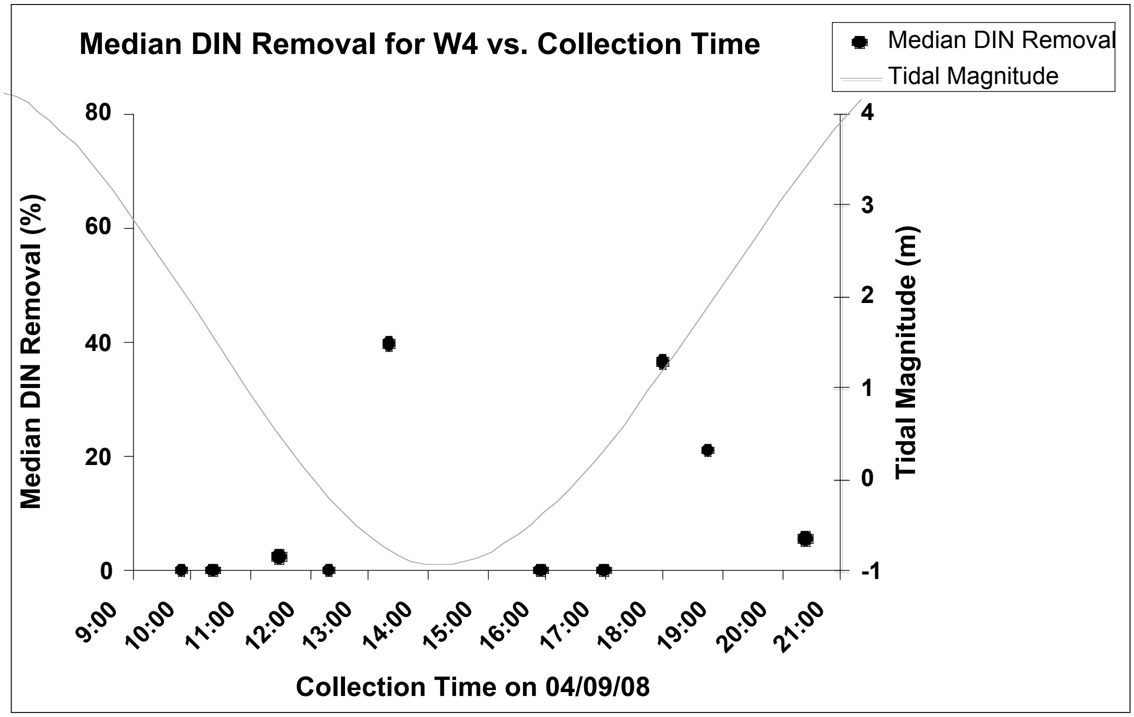


Figure 29. MAS05 experiment #1 median DIN removal percentages calculated for each sample collected from well 4 plotted against collection time. Median removal values were highly variable and did not correlate with the tidal cycle.

The DIN removal probabilities for wells up-gradient and down-gradient of the bulkhead are shown in Figures 30 and 31, respectively. The distributions of DIN removal probabilities for the two data sets were nearly identical, although there was a slightly greater probability of conservative N transport to wells UGBH than those DGBH (71% and 60%, respectively). Although the bulkhead is a physical barrier, it does not appear to have an effect on the transport and fate of N.

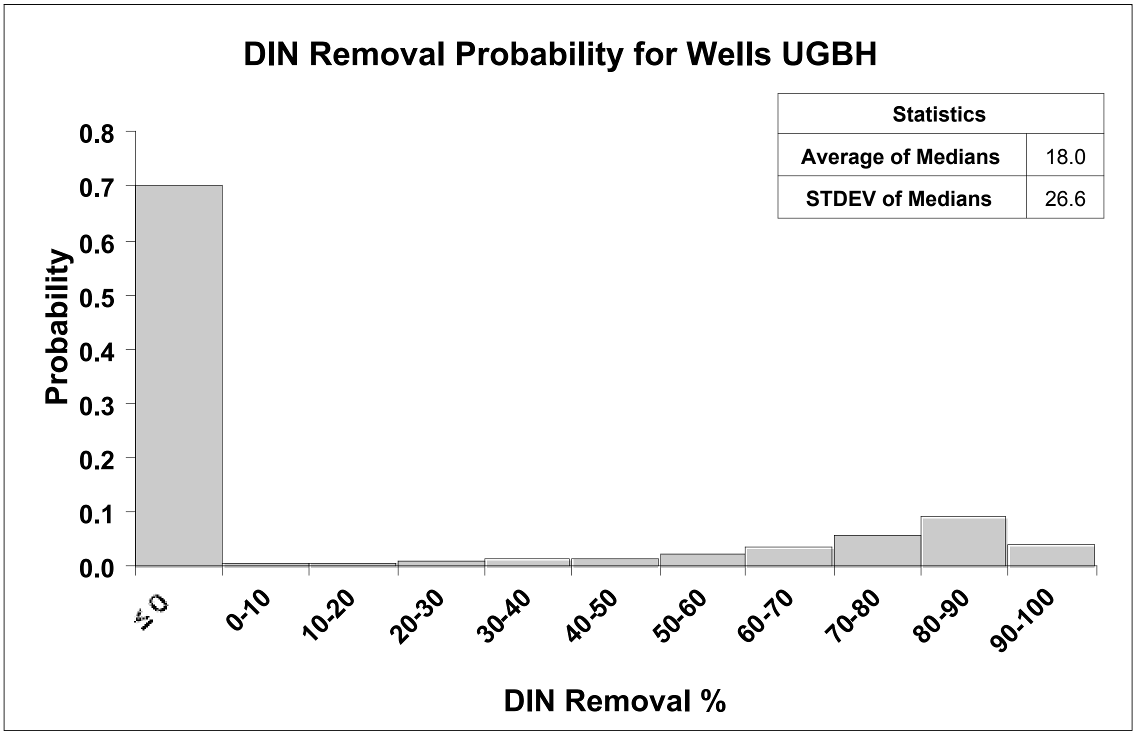


Figure 30. MAS05 experiment #1 DIN removal histogram pertaining to wells located UGBH. The data are skewed towards conservative transport of N. The probability that the N in OSS effluent was conservatively transported to these wells was 71%.

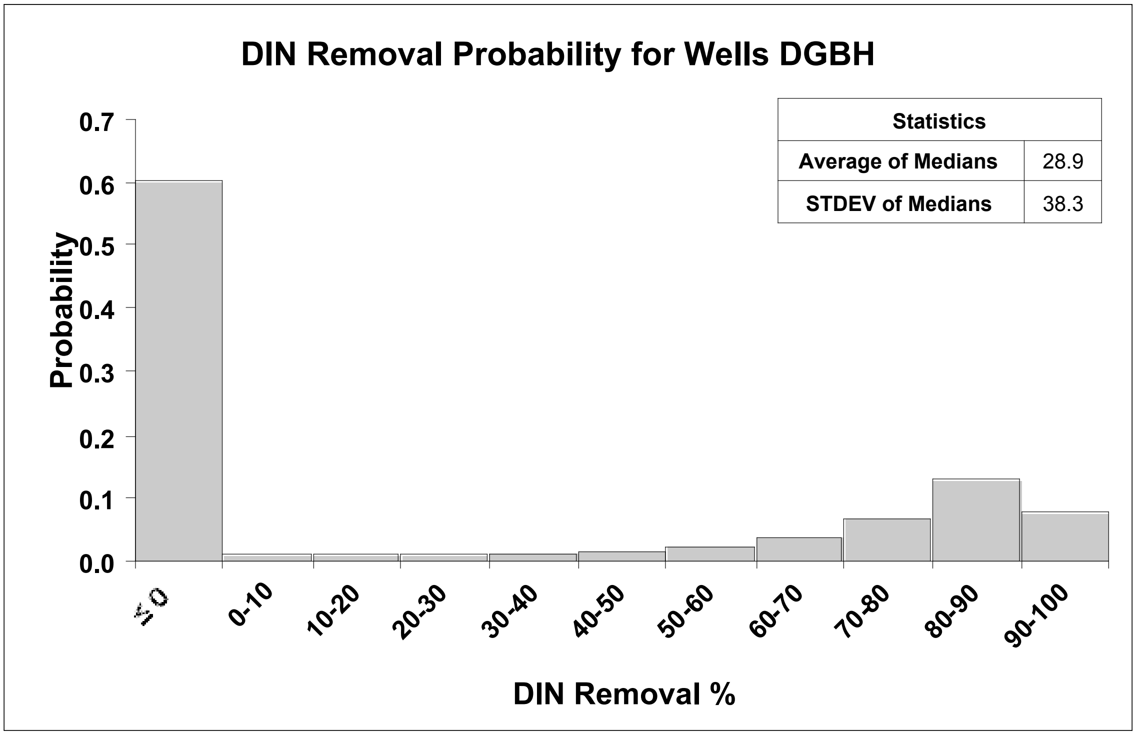


Figure 31. MAS05 experiment #1 DIN removal histogram pertaining to wells located DGBH. The probability that the N in OSS effluent was conservatively transported to these wells was 60%. The distribution of N removal probabilities for wells DGBH is very similar to that of wells UGBH.

3.6.2 MAS05 Experiment #2

The second experiment conducted at MAS05 produced the most complete data set. The key difference between the first experiment and this experiment was that wells adjacent to the drainline were sampled during the second experiment. The DIN removal probabilities based on all samples adjacent to the bulkhead are displayed in Figure 32. Similar to the results of the first experiment, the data indicated a strong likelihood that little or none of the N released from the OSS was removed by the time it reached the sampling wells, although the computed 44% probability of zero removal was somewhat less than in the first experiment. The mean DIN removal levels computed for individual wells are similar to those of Well 4 from the first experiment, in that they were scattered and did not correlate with the tidal cycle.

Similar to the first experiment, the differences between the UGBH and DGBH data sets were minimal. The distributions of DIN removal probabilities associated with each data set were nearly identical, with a slightly greater probability of conservative N transport to wells UGBH than those DGBH. The DIN removal probabilities associated with wells adjacent to the drainline indicated conservative transport, as expected. Thus, N removal appears to be limited throughout the system.

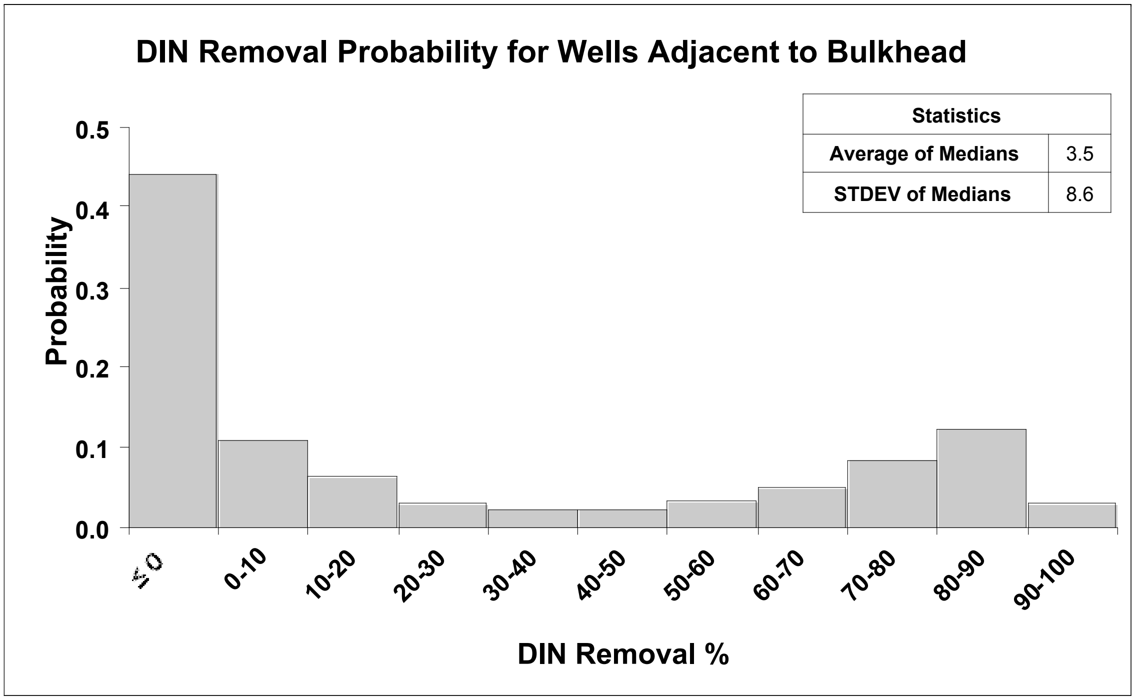


Figure 32. MAS05 experiment #2 DIN removal histogram for wells adjacent to the bulkhead. The data are skewed towards conservative transport of N. The probability that the N in OSS effluent was conservatively transported to the wells adjacent to the bulkhead was 44%.

3.6.3 MAS07 Experiment #3

A comprehensive assessment of DIN removal probabilities for all samples collected adjacent to the bulkhead is displayed in Figure 33. The data indicate a 72% likelihood of conservative transport of N to the bulkhead. The DIN removal probabilities associated with wells adjacent to the drainline were even more skewed towards conservative transport, as expected. Note, however, that only thirteen samples from five different wells were collected over the course of the experiment. The low number of samples limits the ability to draw conclusions, since such conclusions would be biased towards the experiment-specific samples and conditions.

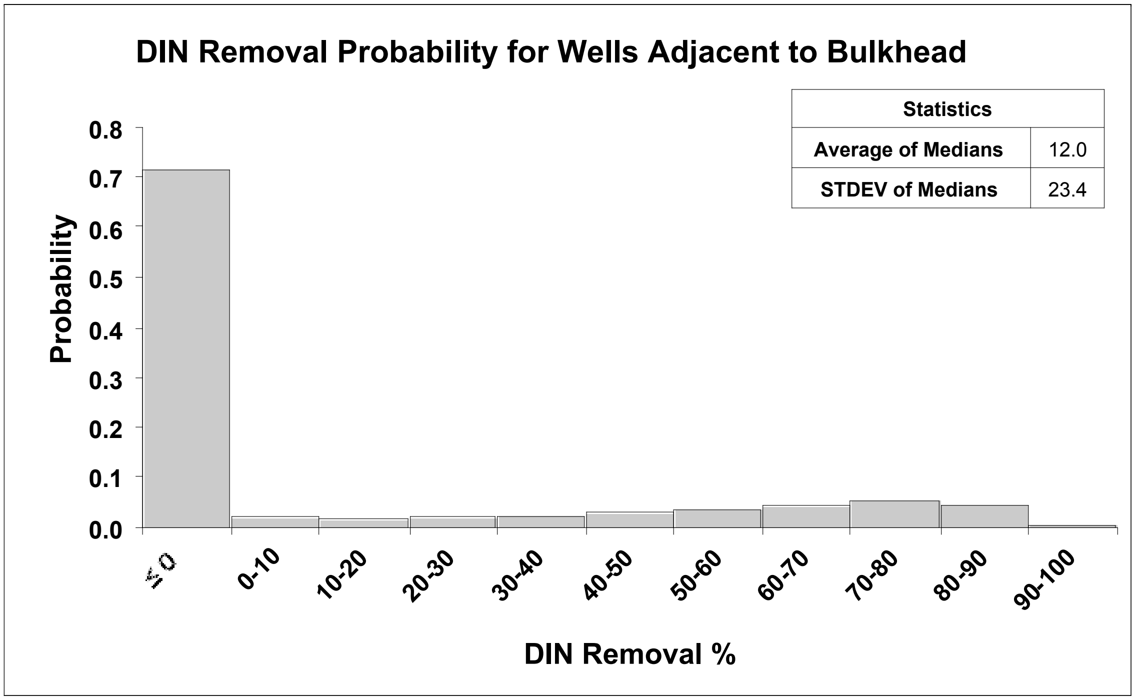


Figure 33. MAS07 experiment #3 DIN removal histogram for wells adjacent to the bulkhead. The data are highly skewed towards conservative transport of N. The probability that the N in OSS effluent was conservatively transported to the bulkhead was 72%.

3.7 Shoreline Sites Discussion

Overall, the experimental results indicate that conservative transport represents the most likely fate of N released from the near-shore OSSs at both MAS05 and MAS07. The likelihood that all N in OSS effluent is conservatively transported to Hood Canal was calculated as 55% for MAS05 and 72% for MAS07. Although some removal of N via denitrification may occur, it appears that such removal is minimal. The median levels of N removal calculated for MAS05 and MAS07 were only 7% and 12%. Both values are significantly lower than the aggregate 35% reduction used for the USGS estimate of the annual N load from subsurface flow discharged by shallow OSSs along Hood Canal (Paulson et al., 2006). The low probability that denitrification occurs in the subsurface down-gradient of the OSSs investigated could be used to improve future N loading models.

The short hydraulic residence time (HRT) and oxygenation of OSS effluent in the soil column caused by seawater infiltration likely limits N removal in the subsurface down-gradient of the OSSs studied. Limitation of N removal by very short HRTs in drainfield soils is plausible because time-requiring processes such as nitrification need to occur before denitrification can proceed. The short HRT of OSS effluent in the systems studied is attributed to the proximity of the OSSs to Hood Canal (a linear distance of ~6.5 m) and to the diurnal infiltration of seawater into the soil column and the consequent flushing of OSS effluent into the Canal. It is unlikely that denitrification will occur after OSS effluent interacts with infiltrating seawater, since such seawater is oxygenated. The average DO concentration measured in near-shore seawater samples was 10.7±0.4 mg/L. Mixing of OSS effluent with infiltrating seawater will instantaneously oxygenate the effluent and thus prohibit denitrification. Therefore, it is reasonable to conclude that the N released from OSSs at sites similar to MAS05 and MAS07 (i.e. minimal gradient, shallow flow path, adjacent to northern shoreline) is minimally removed before entering Hood Canal.

A number of samples collected from wells up-gradient of the bulkheads comprised >99% seawater, indicating that the bulkheads at both sites, but particularly MAS05, do not hinder seawater infiltration. Furthermore, the pressure transducer data collected at both sites indicate that seawater regularly infiltrates the subsurface down-gradient of both OSSs. Finally, no significant differences distinguish the N removal values associated with wells UGBH from those DGBH at MAS05. Thus, although the bulkhead is a physical barrier for surface waves, it does not appear to affect the transport and fate of N and other dissolved constituents. Therefore, once OSS effluent reaches the boundary of seawater infiltration, it can freely mix with infiltrating seawater and be discharged into the canal. The influence of OSS effluent was evident in a number of samples based on elevated DIN concentrations, as infiltrating seawater that had presumably mixed with effluent retreated back to Hood Canal.

The N removal levels calculated for the OSSs studied can be used to approximate the N load from those OSSs and similar systems. However, a number of properties along the shoreline of Hood Canal differ from the sites examined in this study, so it is not necessarily appropriate to extrapolate results from this study to all shoreline sites. As previously discussed, denitrification activity varies spatially. The fact that experiments were conducted at two sites in close proximity and with similar subsurface characteristics makes it likely that only a narrow range of denitrification levels were observed. The removal of N from OSS effluent released at other properties could significantly differ from the degree of removal observed in this study. In particular, OSSs on properties with a steeper grade and deeper flow paths may permit more N removal. The longer HRT of effluent in the subsurface due to longer flow paths and decreased seawater infiltration could lead to increased N removal.

3.9 Shoreline Sites Conclusions

The fate and transport of anthropogenic nitrogen N to coastal marine systems is important and warrants further investigation. Nitrogen loading from OSSs to such systems, specifically the Hood Canal, could have negative ecological impacts. Conventional OSSs at two sites located immediately adjacent to the shoreline of southern Hood Canal were investigated in this study. Three field experiments were conducted to examine the fate and transport of N released from these OSSs. These results indicated that conservative transport represents the most likely fate of N released from the OSSs at both study sites. Although results demonstrated that some denitrification occurred down-gradient of near-shore OSSs, the extent of denitrification was minimal. The results of the two experiments conducted as MAS05 were generally in agreement and supported the hypothesis of conservative transport of N. The results of the experiment at MAS07 also indicated conservative transport. The short hydraulic residence time and oxygenation of OSS effluent in the soil column caused by the semi-diurnal infiltration of seawater likely limits N removal via denitrification at the experimental sites. Data indicated that infiltrating seawater prematurely terminates the flow paths of OSS effluent. The results and subsequent conclusions of this study probably apply to OSSs similar to those investigated. However, extrapolation of results to OSSs with differing characteristics is problematic. There could potentially be a wide range of N removal levels occurring in the subsurface down-gradient of OSSs located along the shoreline of Hood Canal. The results of this investigation may only represent a small range of actual values since denitrification activity varies spatially and temporally. Further research should address the limitations of this study with respect to the tracer used, the low number of study sites and the seasonal nature of experiments conducted.

4. Mason 10

4.1 Introduction and Goals

The MAS10 site is located in Belfair (Figure 34). Shallow wells were installed at the site in the fall of 2007. The shallow wells did not extend to the groundwater table beneath the site. Artificial rainfall experiments were performed throughout the fall and winter of 2007. The wells and septic tank were sampled to explore N transformations, and a probabilistic model was used to estimate removal near the surface. Deep wells, extending into the groundwater table, were installed in March 2008 and routinely sampled until August 2008. The deep wells, septic tank and background wells were sampled, and the probabilistic model was used to estimate N removal in the deep wells.

Figure 34: Hood Canal, with Belfair shown at the terminus.

4.2 Mason 10 Site Description

The MAS10 site is a trapezoid-shaped property covering 0.7 acres. The ground surface slopes gently down towards Lynch Cove, approximately 0.4 km away. Site geology is mapped as Quaternary advanced outwash (Dragovich et al., 2002). Groundwater is approximately 4 to 5 meters below ground surface (bgs), based on borings described later. Surrounding properties include residences and a large public facility (PF) upslope from the site. A vicinity map showing the site and surroundings is shown in Figure 35. (Note: the description of the surroundings and the vicinity map are intentionally unspecific to protect the confidentiality of the site owner.)

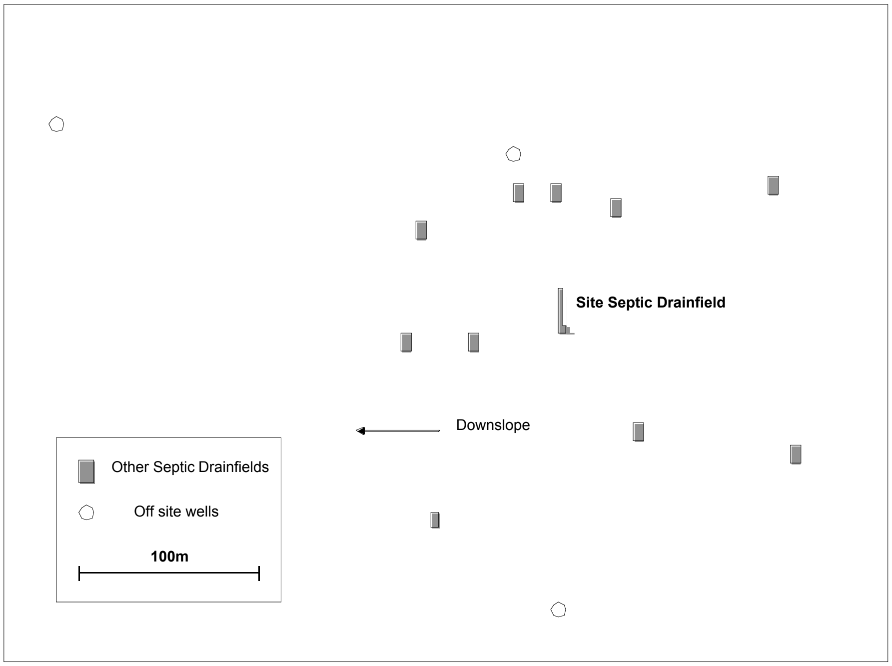
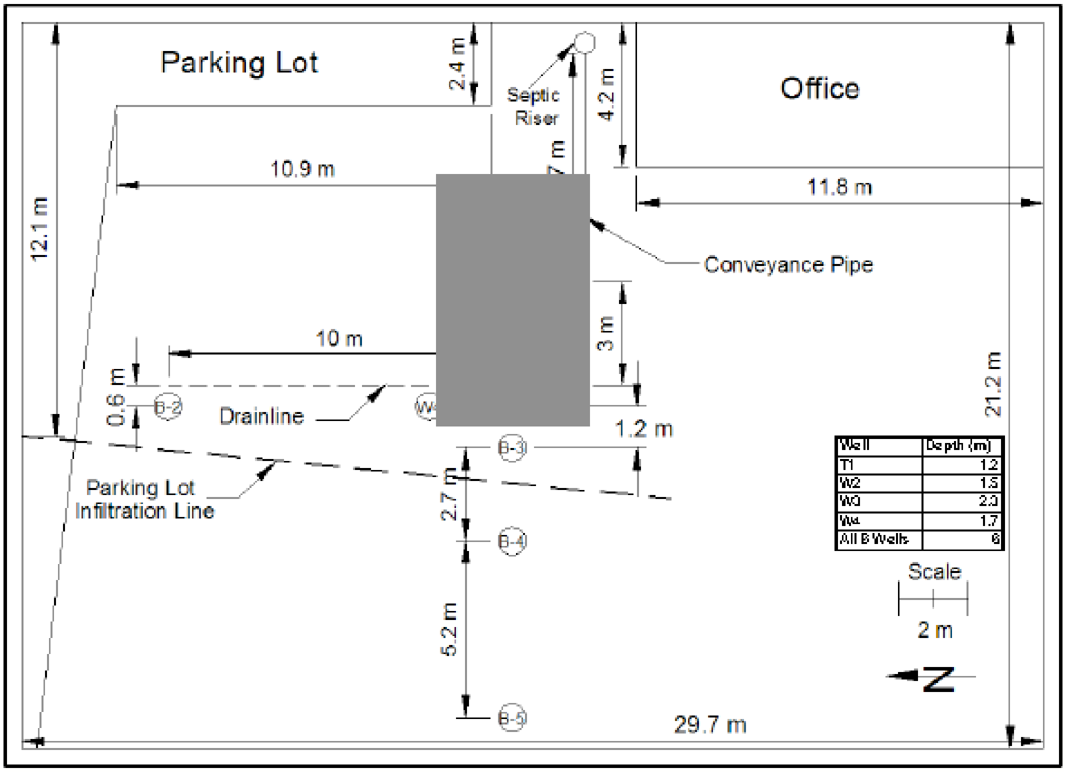


Figure 35. MAS10 vicinity map, showing the relative location of off-site wells and other known and suspected drainfields.

The site is zoned for commercial use and contains a small office building. The building was expanded from 55 m^2 to 130m^2 in 1999. At that time, the septic tank was also replaced, and the drainfield was excavated and repaired in places. Typical staffing at the site is 5 to 7 employees, Monday through Friday, 8:00 to 17:00. Staffing levels tend to be lower in the winter and higher during the remainder of the year. The site contains asphalt paved parking for approximately 12 vehicles in a parking lot that slopes toward a single catch basin, which collects stormwater and directs it via a buried 10 cm diameter PVC pipe to a lawn west of the building. Stormwater runoff from the parking lot and a portion of the adjacent street infiltrates in site soils under the lawn.

Wastewater at the site is treated by a conventional gravity OSS. Wastewater is generated within the building from a single bathroom with one toilet and sink, as well as from one sink in a kitchenette. The sink in the kitchenette does not have a garbage grinder. Water meter readings recorded approximately every other week from March until July 2008 indicated site water use was approximately 1,200 liters per week. These readings were obtained before landscapers began using water for irrigation in the summer. Most irrigation water was used in areas of the site that are not near the OSS drainfield.

Flows from within the building are directed to a 5,700 L concrete septic tank located below ground, approximately 1.5 m north of the building (Figure 36). The OSS as-built drawing from the 1999 construction indicates the septic tank does not contain baffles. The drawing also indicates that wastewater exits the septic tank and flows via a buried conveyance pipe 6.1 m to the west. An L-shaped infiltration trench, approximately 12 m long and 1 m wide, is located under the lawn. The installer for this system recalled that the infiltration trench depth was probably 0.5 to 0.7 m. The infiltration pipe from the catch basin in the parking lot is parallel to the 12 m section of the OSS infiltration trench, and the two pipes are approximately 1 to 2 m apart (Figure 36).



Well	Depth (m)
T1	1.2
W2	1.5
W3	2.0
W4	1.7
All B Wells	6

Figure 36. MAS10 site plan. Wastewater flows from the building into a septic tank (riser shown), then west to an infiltration trench. A single catch basin collects stormwater from the parking lot and directs it via buried 10 cm diameter PVC infiltration line into the lawn. The wells are described in Section 4.3. The shaded area is the irrigated area described in Section 4.3.

Visual examination of soil cores collected from borings on March 6, 2008, indicate the site is underlain by approximately 0.6 to 0.9 m of fill, comprised of sand with silt. Approximately 1.2 to 1.8 m of native sand with gravel and silt was observed underneath the fill. Native gravel with sand and silt was observed beneath this layer to the full depth explored, which was 6 m in all borings. The presence of fill soil was marked by a soil color change and by an increase in soil density (based on conversations with the drillers operating the drill rig) below 0.6 to 0.9 m. Furthermore, while layering was not observed in the upper 0.6 to 0.9 m of the borings, soil layers resulting from grain size changes were observed beneath the 0.6 to 0.9 m depth. Observations of site soils are consistent with the geologic map of the northwest quadrant of Washington State (Dragovich et al., 2002), which classifies the soils in the area as advanced outwash. Groundwater was encountered approximately 4.25 m bgs in all borings on the day of drilling. Figure 37 shows soil characteristics in cross-section through borings B-1, B-3 and B-4.

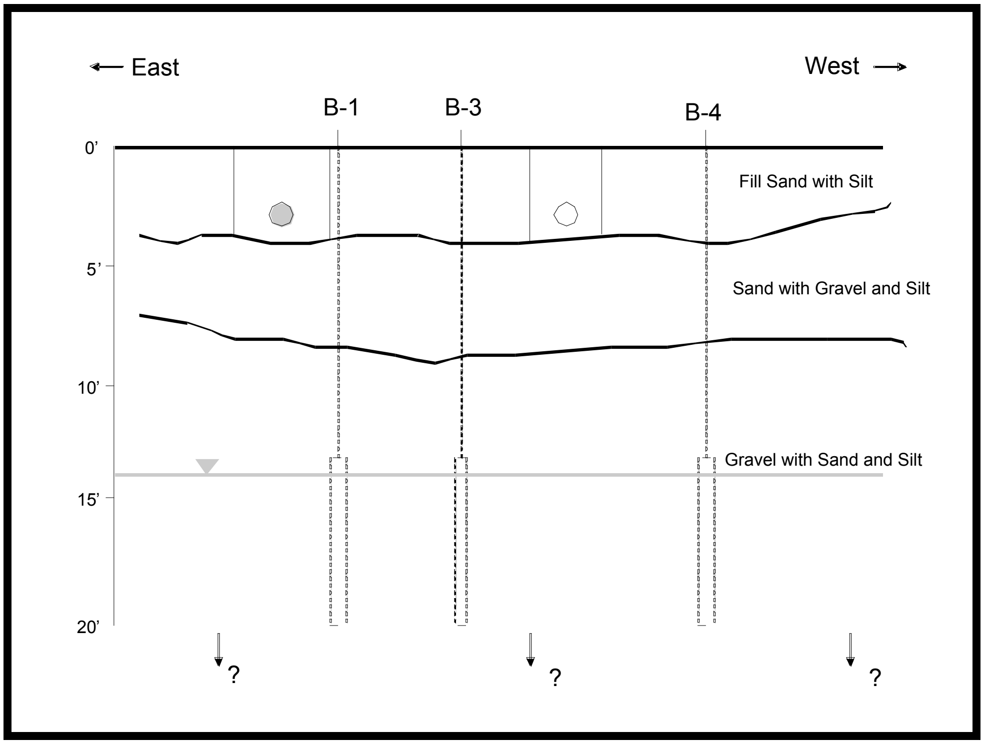


Figure 37. Cross section through B-1, B-3 and B-4. Scale shown on left side. No vertical exaggeration. Boring locations described below. Drainfield trench on left, trench for parking lot infiltration line is between B-3 and B-4. B-5 was not soil sampled and is therefore not shown.

4.3 Mason 10 Site-Specific Methods

The general methods describing well installation and sampling are found in Section 2. Methods unique to the MAS10 site are described below.

4.3.1 General

Work at MAS10 can be divided into two periods, an "irrigation only" period and a "routine sampling" period. During the first period, from August until December 2007, shallow wells were installed as described in Section 2 near the drainfield, and four irrigation experiments were conducted. The shallow wells extended down to a maximum of 2.3 m bgs and did not penetrate the groundwater table beneath the site. The irrigation experiments were used both as a means to facilitate sample collection and to address the issue of nitrogen storage and flushing.

The second period began in March 2008, when six deep wells were installed at the site using a commercial drill rig. These deep wells were installed into the groundwater table beneath the site, enabling routine sampling without irrigation. Furthermore, one of the shallow wells (T1) installed during the "artificial rainfall only" period was found to contain water even without irrigation and was sampled along with the deep wells during routine sampling. A fifth and final irrigation experiment was performed in June 2008, when both shallow and deep wells were sampled.

The conservative tracer Br^-, was added to the septic tank once during the first period of the study, and twice during the second period. On September 20, 2007, four days before the first irrigation experiment, 100 g of Br^- was added in 1 L of solution. On March 31, 2008, 25 days after the deep wells were drilled, 342 g of Br^- was added in 20 L of solution. Finally, 2,260 g of Br^- was added in 20 L of solution on May 14, 2008. The decision to increase the Br^- dose dramatically during the final addition was made because Br^- was detected at only very low concentrations in deep wells after the March 31, 2008 dosing.

4.3.2 Wells

Fifteen shallow wells (T1, T2 and W1-W13) were installed at the site manually in August 2007. Most of the shallow wells never yielded water, even during irrigation experiments. The locations of the shallow wells that consistently yielded water are shown in Figure 38, along with deep wells (described below). Shallow well depths ranged from 1.2 to 2.3 m bgs. Shallow wells were the 1.27 cm diameter wells described in Section 2. T1was constructed of 2.54 cm diameter PVC for accommodating a pressure transducer.

4.3.3 Irrigation

Five irrigation experiments were conducted at the site, on September 24, October 29, November 15 and November 29, 2007 and June 18, 2008. During each experiment, one or two lawn sprinklers were used to irrigate either directly over the drainfield trench or adjacent to it for 9-20 hours. The area irrigated during experiments 1, 4 and 5 is shown in Figure 36. The areas watered during experiments 2 and 3 were to the west of the area indicated in Figure 36, however very few wells contained water during these experiments. Therefore, only the data from experiments 1, 4 and 5 are discussed in this section of the report. Samples were collected hourly from the time water entered the shallow wells (typically 5 to 9 hours after the start of the irrigation) for approximately 8 hours. Samples were then collected approximately every four hours until approximately 24 to 36 hours after the start of irrigation. The deep wells were available for sampling during experiment 5, and samples were collected from deep wells every hour from 9:00 until 17:00 on June 18, then approximately every four hours until 8:00 June 19. Table 1 summarizes the experimental conditions.

Table 8. Irrigation Experiment Summaries

Experiment Number	Date	Liters Water Used	Irrigation Time (hours)	Irrigation Area (m^2)	Equivalent cm of Rain
1	9/24/2007	26000	28	34	76
2	10/29/2007	25000	25.5	40	61
3	11/15/2007	12000	19	28	46
4	11/29/2007	15000	24	28	53
5	6/18/2008	8700	17.5	34	25

4.3.4 Drilling and deep well installation methods:

Six deep wells (B-1 through B-6) were installed on March 6, 2008, using direct push technology from a drilling company under contract to the University of Washington (see Figure 36 for well locations). The drilling equipment included a Geoprobe® 54DT

Megaprobe® mounted on a 2-ton pickup truck. Each boring except B-5 was first continuously soil sampled using a 1.2 m long, 3.2 cm outer diameter Geoprobe® soil sampler and disposable acetate liners. Well B-5 was not soil sampled due to time constraints. Wells B-1, B-3, B-4 and B-5 were located in a line going away from the known OSS "hot-spot" of T1, in an attempt to measure dilution of the OSS plume. Unfortunately, no wells could be installed east of B-1 in the vicinity of the drianfield trench because the exact location (i.e. north and south) of the pipe coming from the septic tank was not known. Well B-2 was installed near the end of the trench to attempt to assess whether or not OSS water was being discharged from the entire length of the trench. Well B-6 was installed as a far as possible from the drainfield as a "background" well.

All deep wells were drilled to 6 m bgs. Groundwater was encountered at approximately 4.25 m bgs on the day of drilling. In all wells except B-5, the soil sampler was driven in 1.2 m increments, and after each run the soil sampler was removed from the boring, and the acetate liner containing the soil column was removed for soil logging and sampling. The soil sampler was then cleaned, and a new acetate liner was installed. The sampler was then re-inserted into the borehole, and 1.2 m sections of 2.54 cm diameter casing rod were used to repeat the process to the full depth of the wells.

After driving to 6 m bgs, all equipment was removed from the borehole, and the hole was overdrilled again to 6 m bgs by driving an 8.3 cm outer diameter steel casing with a sacrificial steel cone tip attached to the bottom end of the casing. One pre-packed well, 1.5 m in length and 1.27 cm in diameter, was installed at the bottom of the borehole. The pre-packed wells consisted of 0.25 mm, slotted, schedule 40 PVC well-screen. The PVC well-screen was wrapped with fine stainless steel mesh, and the annular space between the outside of the slotted PVC well-screen and the stainless steel mesh was filled with clean silica sand. The remainder of the well was cased using 2.5 cm diameter, schedule 40 PVC well casing.

After installing the PVC well, the steel casing was slowly removed as clean 20-40 silica sand was poured into the annular space between the PVC well and the 8.3 cm diameter steel casing. The casing separated from the sacrificial drive point, and the drive point remained at the bottom of the borehole permanently underneath the well screen. Sand was poured to fill the space from approximately 4 to 6 m bgs, i.e., to a depth of 0.6 m above the top of each well screen. Bentonite chips were poured into the annular space between the PVC well and casing as the remainder of the casing was slowly removed. Bentonite chips were used to fill from approximately 0.6 to 4 m bgs.

Between the ground surface and approximately 0.6 m bgs, each well was set in steel cylindrical monuments measuring 12.7 cm in diameter and 31 cm in depth. Portland cement was hand-mixed onsite and used to set the monuments, the tops of which were mounted approximately flush with the surface of the lawn.

The exception to the description above was the well B-6. When installing this well, drillers encountered dense gravel and cobbles at approximately 4 m bgs while driving the 8.3 cm diameter casing. The casing was removed and the hole was re-drilled to 6 m bgs using a 3.8 cm diameter drive point. The drive point was removed and a 1.9 cm diameter PVC well screen was installed. The well-screen consisted of a 1.5 m length of 0.25 mm slotted schedule 40 PVC. Although the well-screen was not pre-packed as in B-1 through B-5, silica sand was poured into the annular space between the well-screen and surrounding soil to create a filter. The remainder of the well construction proceeded similarly to that for B-1 through B-5.

4.3.5 Autosamplers

Two Teledyne Isco® 6712 Portable Samplers were also used at the site to collect samples from the septic tank and from B-1 between site visits. The samples were taken to the laboratory after each site visit and analyzed for Br^- to monitor concentrations in the septic tank and to detect Br^- “breakthrough” in B-1.

4.4 Probabilistic Model Overview

4.4.1 General

The probabilistic model used during Year 1 (Horowitz et al., 2007) was used to assess N removal in and down-gradient of each OSS drainfield. An attempt was made to account for the uncertainty associated with variation in OSS loading due to temporal variations in both flow and composition of the wastewater.

4.4.2 Measures and Assumptions of Nitrogen Speciation

Nitrogen speciation in the OSS effluent was determined directly from the measured data. As described in Section 2, well and OSS samples were analyzed for NO_3^-, NH_4^+, and TDN. For the well samples, the TDN concentration was assumed to be the sum of the nitrate and ammonium concentrations; i.e., organic nitrogen was assumed to be negligible. For the septic tank samples, the TDN concentration was measured directly.

4.4.3 Data Analysis Tools for Determining TDN and DOC Removal

The probabilistic model developed in the Year 1 report was used to estimate the percentage removal of TDN and DOC in the OSS effluent. The estimated removals reflect the cumulative effect of processes occurring between the septic tank and the sampling location and therefore do not necessarily indicate whether TDN or DOC removal is occurring in the immediate vicinity of the sampling location. Samples with Cl^- concentrations less than one standard deviation above the background concentration were excluded from the model analysis. In these cases, it is assumed that the OSS plume constituted a small portion of the water sample, in which it is difficult to predict an expected TDN concentration.

4.4.4 Data Analysis Tools for Assessing DOC Limiting Denitrification

The stoichiometry of the denitrification reaction can be used to evaluate whether sufficient DOC is available to denitrify all of the TN in the water. A model reaction showing this stoichiometry is as follows (Crites and Tchobanoglous, 1998):

$$C_{10}H_{19}O_3N + 10NO_3 \rightarrow 5N_2 + 10CO_2 + 3H_2O + NH_3 + 10OH \qquad \text{(Equation 8)}$$

Based on this reaction, 0.86 g of C is required to denitrify 1.0 g of N. However, not all DOC is degradable, and the composition of the DOC might differ from that of the model molecule specified above (in which the average oxidation state of carbon is −1). Based on these considerations, Korom (1992) suggested that a ratio of approximately 1 g DOC/g TDN or greater indicates that sufficient DOC is present to reduce the available TDN. Correspondingly, we assumed that a ratio of approximately 1 g DOC removed per g TDN removed is consistent with a hypothesis that the DOC provided the electrons necessary for denitrification.

4.5 Mason 10 Results

4.5.1 Septic Tank

Analyses of 20 septic tank samples and 15 T1 samples collected between September 2007 and July 2008 are presented in Table 3. Literature wastewater data and data compiled from four other septic tanks in the Hood Canal watershed are also shown in the table. The concentrations of both TDN and Cl^- are higher in the septic tank than in typical untreated domestic wastewater (Crites, 1998) and in four other septic tanks in Hood Canal (Horowitz, 2008), because domestic wastewater contains a larger percentage of greywater than does the site's wastewater (there are no shower or laundry facilities at the site). Greywater dilutes the TDN and Cl^- in blackwater. Water collected from T1 is strongly influenced by OSS effluent. Only DOC and BOD are significantly reduced in T1 compared to the septic tank.

Table 9. Septic tank data for MAS10 and others, and T1 data.

	TDN (mg/L)	DOC (mg/L)	Cl (mg/L)	BOD (mg/L)[1]
MAS10 Average	119	62	81	177
Site SD	20	17	28	17
Crites "Typical"	35	160[2]	50	210
Crites Range	20-85	80-290[2]	30-100	110-400
Hood Canal Average	54	64	22	NA
Hood Canal SD	8	12	10	
T1 Average	94	22	67	61
T1 SD	24	7.0	11	15
Percent Reduction[3]	21%	65%	17%	66%

[1] Five day, twenty degrees C
[2] Crites gives total organic carbon, not dissolved organic carbon
[3] Between the septic tank and T1
NA = Not available

Chloride, DOC and TDN concentrations in the tank decreased at the beginning of the study, but were variable after March 31 (Figure 38). Additionally, the TDN:Cl ratio increased by 60% (from 1.26 to 1.92) over the course of this study (Figure 39). There are two possible explanations for the concentration changes: 1) Changes in the number of employees in the office, or changes in employee practices, may have occurred. The site manager indicated that staffing is typically higher in the spring and summer and lower in the fall and winter. 2) The timing of pumping of the septic tank may have affected dissolved constituent concentrations. The site owner reported that the septic tank is pumped approximately every two years, but could not recall the last date of pumping. The depth of the sludge layer was approximately 2 to 5 cm on September 20, 2007, which suggests that pumping occurred not long before the study began. The tank was not pumped during the study.

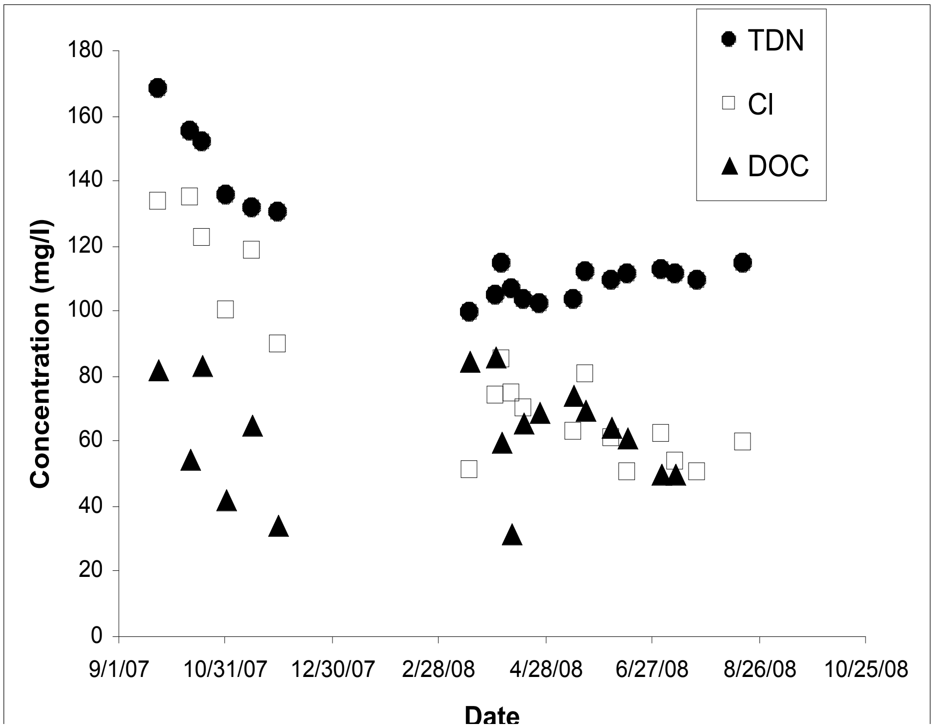


Figure 38. DOC and Cl⁻ concentrations of septic tank throughout the study.

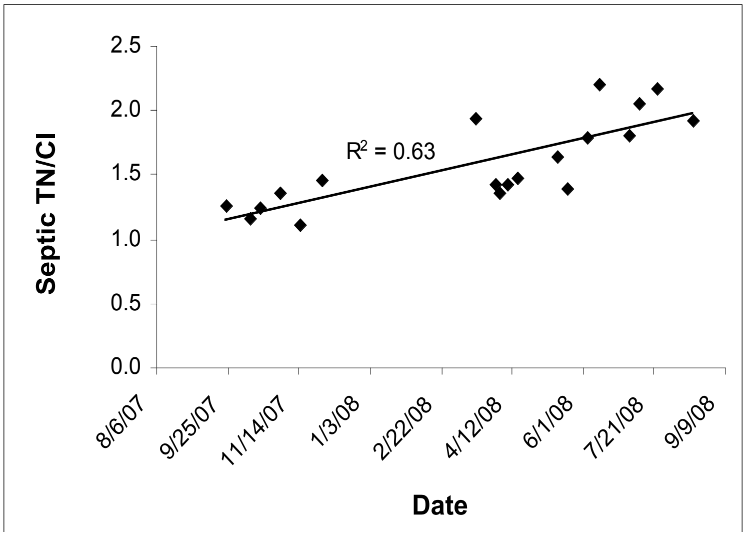


Figure 39. Septic tank TDN:Cl ratio increased throughout the study.

4.5.2 Deep Wells

4.5.2.1 Soil

The sum of NO_3^- + NH_4^+ in soil samples collected from B-1 was 20 ± 2.8 mg/L, and for all other wells the sum was 2.9 ± 2.3 mg/L. The samples were collected from depths ranging from 0.5 to 4 m bgs. The high concentrations observed in B-1, but not other wells, is consistent with a downward-moving plume emanating from near B-1.

4.5.2.2 Groundwater

Samples collected from the deep wells could provide evidence in support of either of two competing hypotheses: 1) Groundwater beneath the site is influenced to a great extent by the site sewage system and, 2) Groundwater beneath the site is influenced to a great extent by a(n) off-site (presumably OSS) source(s). The arguments supporting these "site-source" and "off-site-source" hypotheses are presented below.

Results of NO_3^- and Cl^- analyses for wells B-1 through B-5 through time are plotted in Figure 40, and average concentrations at B-1 through B-6 are shown in Figure 41. Ammonium was not detected in any deep well samples. Results for B-6 are not included in Figure 40 because the well did not yield water after June 18, most likely due to a combination of a falling water table and a poor filter pack in B-6 (described in Methods above). The results shown are from 17 site visits made between March 13 and August 18, 2008. Results from the intensive sampling during irrigation experiment 5 are treated separately below. Nitrate and Cl^- concentrations generally increased during the study. Additionally, the average TDN:Cl ratio in the deep wells increased by 98%, from 0.58 to 1.15.

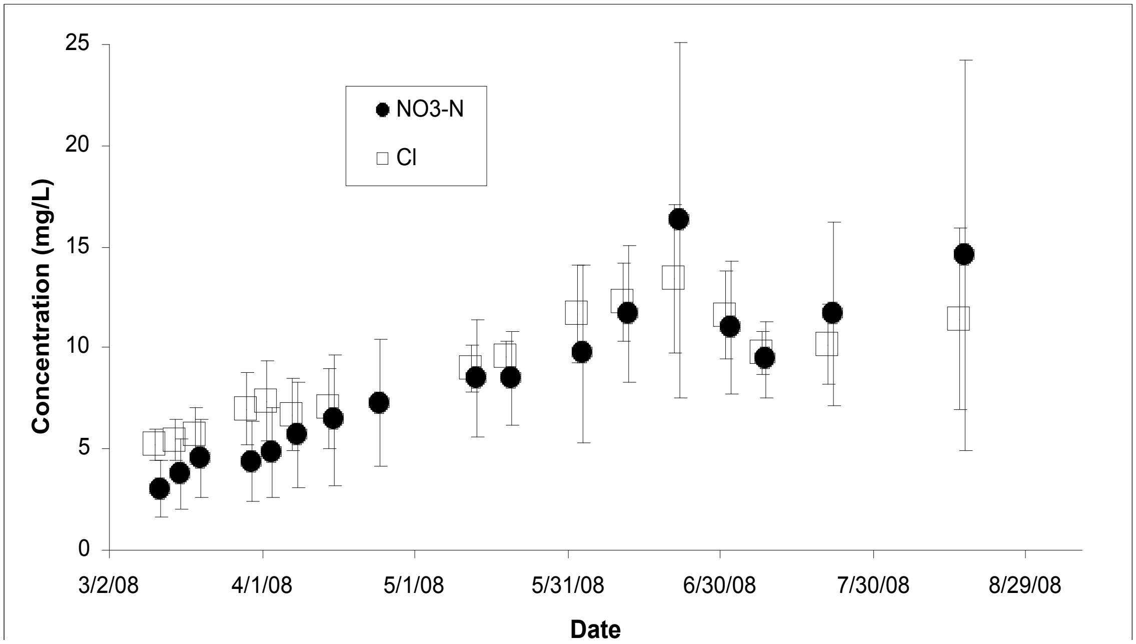


Figure 40. Average NO_3^- and Cl^- concentrations in B-1 through B-5 from March 13 to August 18, 2008. Chloride data is not available for 4/24. Error bars indicate 1 standard deviation. Chloride data set has been offset by one day for clarity. The off-scale values (+1 SD NO_3^- on 6/23 and 8/18) are 25.1 mg/L and 24.2 mg/L, respectively. No Cl^- data are available for 4/25/08.

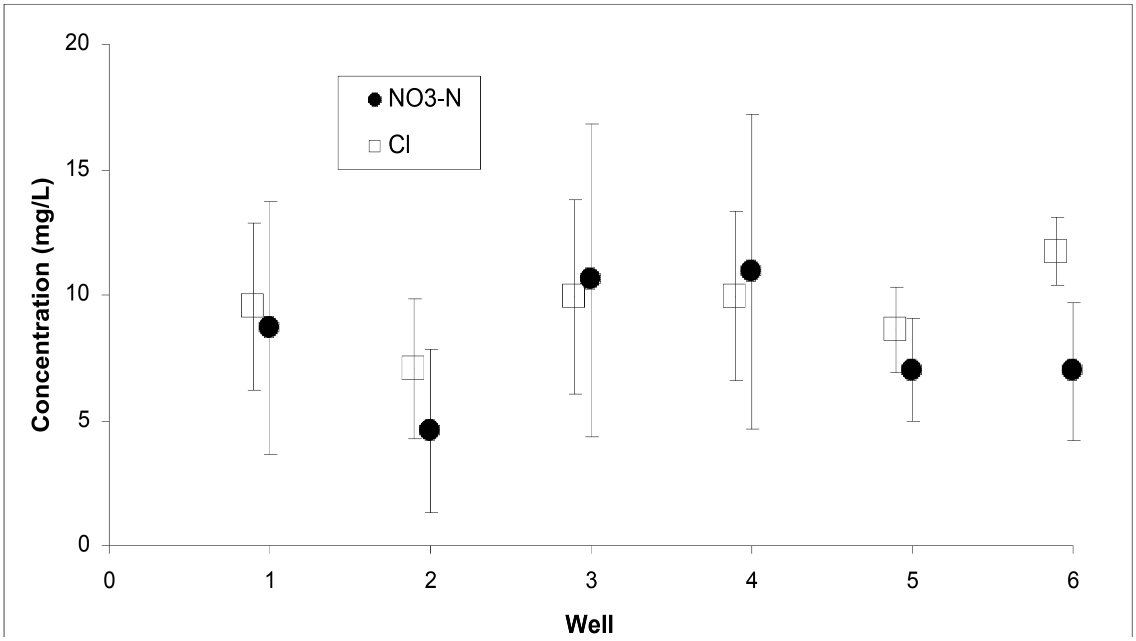


Figure 41. Average NO_3^- and Cl^- concentrations for each deep well. Error bars indicate 1 standard deviation. N=17 for all deep wells except B-6, where N=11.

The concentrations of NO_3^- and Cl^- in the deep wells are within the ranges measured at other sites in the Hood Canal watershed that are influenced by OSS (Horowitz et al., 2007). Nitrate was not detected in the off-site wells (detection limit = 0.16 mg/L), and the Cl^- concentration for the off-site wells was 2.5 ± 0.9 mg/L. Ammonium was detected in two of the three off-site wells, at a concentration of 0.66 ± 0.23 mg/L. The high concentrations of NO_3^- and Cl^- in the deep wells, compared with low background concentrations in the three surrounding off-site wells, support the site-source hypothesis, as do the simultaneous increases of the TDN:Cl ratios in the site septic tank and on-site deep wells. The consistent and substantially different Cl^- concentrations in B-1 through B-5, despite their being located within 13 m of each other, provide additional support for the site-source hypothesis. This is because, while the concentrations of conservative contaminants in any plume typically decrease with distance from the source, the difference in concentrations between any two relatively closely-spaced points (relative to the distant source) should be small.

Depth to water, measured in pressure transducers in B-2 and B-5, fell from 4.31 meters bgs to 5.39 meters bgs between March 18 and August 18 (Figure 42). This offers one possible explanation for why NO_3^- and Cl^- concentrations in the deep wells increased during the study period. Less dilution by groundwater, coupled with a constant, or even increasing, load (due to an increasing population as summer nears) could cause concentrations to increase.

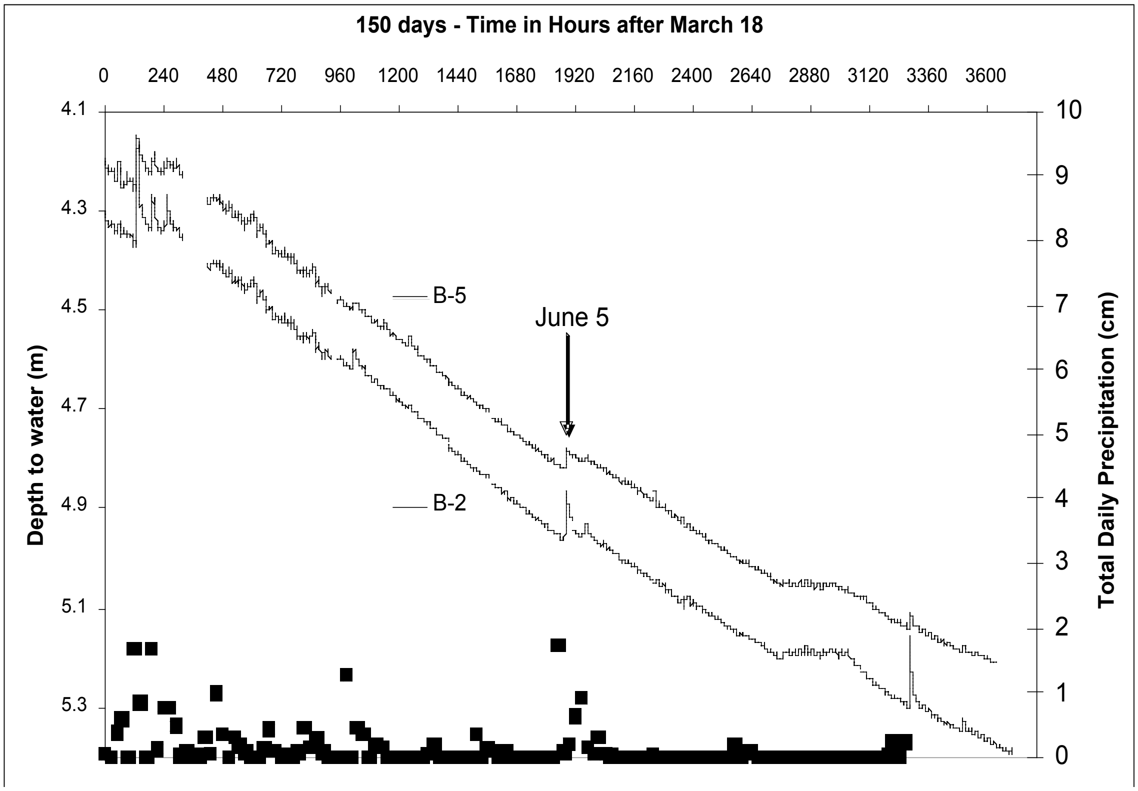


Figure 42. Depth to water in B-2 (top line) and B-5 (bottom) from March 18 to August 18. The two data sets have been artificially separated by 10 cm for clarity. Also plotted along the bottom of the figure are total daily precipitation data (Kitsap PUD) as measured at McKenna Falls, located approximately 10 km north-northeast of the site. The gaps in the pressure transducer data are for times when the transducers were not recording data.

Precipitation data (Kitsap PUD) from McKenna Falls, located approximately 10 km north-northeast of the site, are shown along the bottom axis of Figure 42. Water level “spikes” in the wells, like the one shown in response to precipitation around June 5, coincide with the precipitation data, except near Hours 2650 to 3000 when the water table seems to stall during its fall. This stall might be explained by recharge from precipitation somewhere upgradient of the site. A comparison of the timing of precipitation events and responses indicates that the travel time for water percolating at the site is probably not more than several days. This supports the site-source hypothesis, because it demonstrates hydraulic connection between the ground surface and the water table below.

The response in B-2 was steeper than in B-5, probably because B-2 was affected by the parking lot infiltration line in addition to percolating water from precipitation. The infiltration line passes within a horizontal distance of approximately 2 m of B-2. The depth of the infiltration line in the vicinity of B-2 is not known, but it is probably not more than 0.7 m bgs based on the location of the inlet of the pipe and assuming a 1% slope for the drainline.

On September 14, a garden hose was used to deliver the equivalent of 1.8 cm of rain to the site catch basin over 22 hours. (The volume of water required to do so was calculated based on the area which drains to the catch basin. This area was estimated on October 18, 2007, when rain was falling at the site.) The response to the catch basin test is shown in Figure 43. The response from a real precipitation event (around June 5) is also shown for comparison. Figure 43 demonstrates that B-2 is affected by the infiltration pipe. Results from the catch basin test can also be used to more precisely estimate the percolation rate. The water level began to rise in B-2 approximately 3 hours after the water was turned on. Depth to the water table was 5.51 m on September 14. Given an infiltration line depth of 0.7 m, this indicates a percolation rate for water originating from the catch basin of 1.6 m per hour.

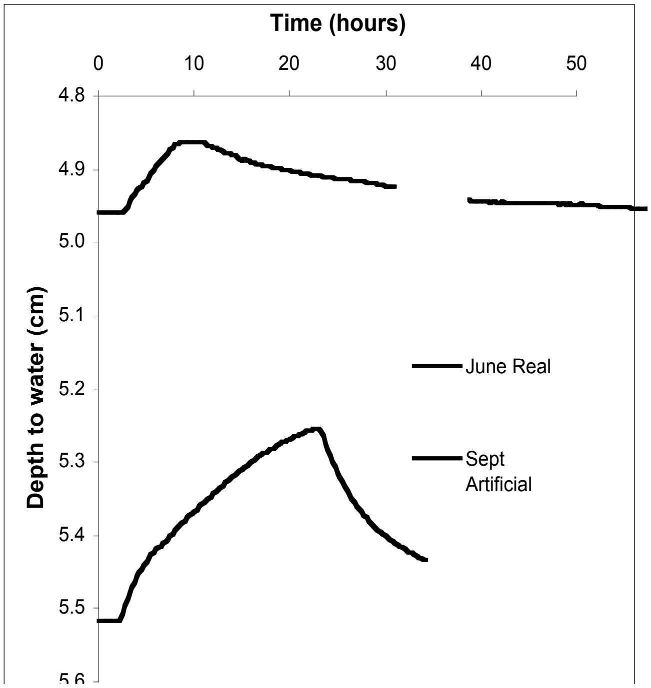
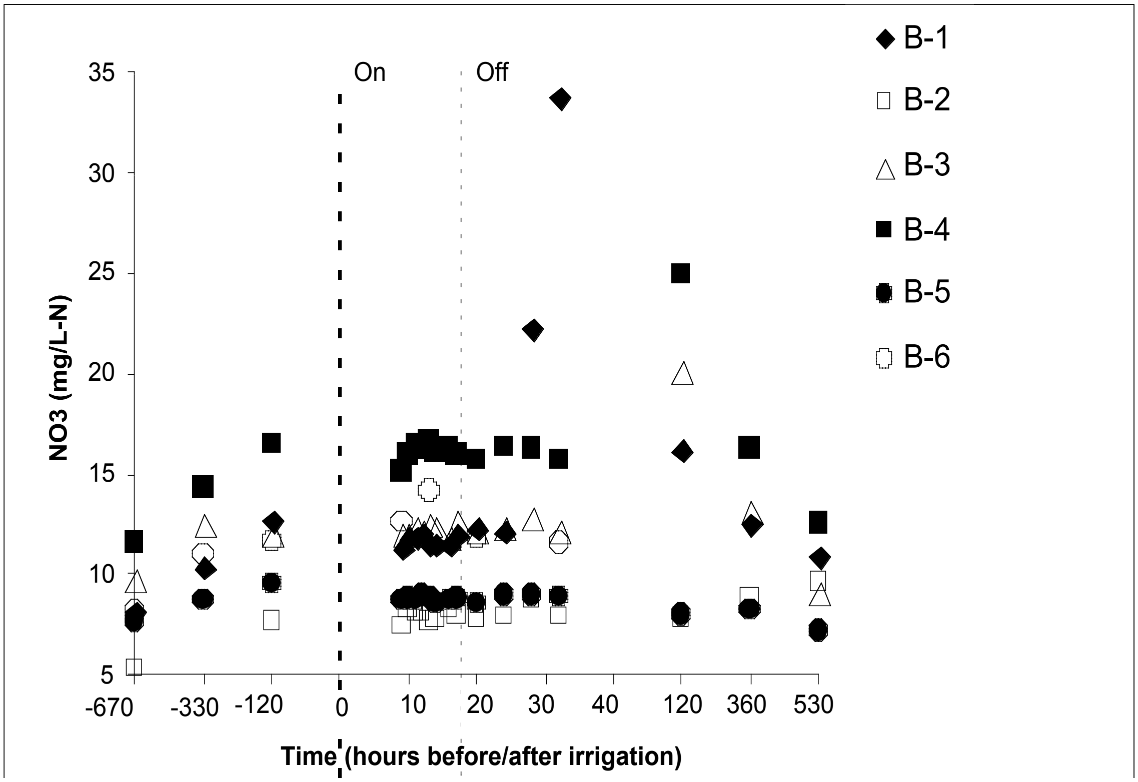


Figure 43. Water level rises in B-2 during an actual precipitation even on June 5 (top line) and during the catch basin test on September 14. There is a data gap in the June 5 data.

During the fifth irrigation experiment, the deep wells were sampled hourly from 9:00 until 17:00 on June 18, 2008, then approximately every four hours until 8:00 on June 19.

Nitrate and Cl^- concentrations in B-1 through B-6 are plotted in Figure 44. Also included in the figure is the data from six routine sampling events, three before and three after the irrigation experiment. Nitrate and Cl^- changed little during most of the experiment, but both analytes were elevated in some wells towards the end and after the experiment.



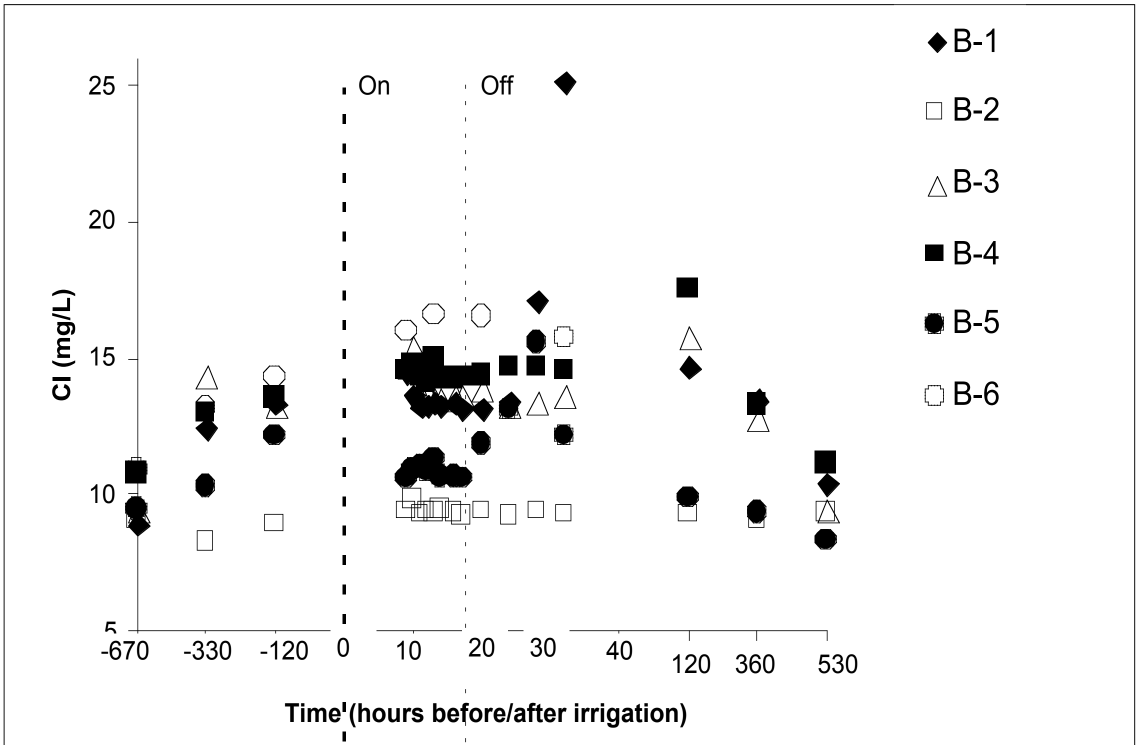


Figure 44. Nitrate (upper) and Cl$^-$ (lower) concentrations in deep wells B-1 through B-6. Time is in hours before or after irrigation was initiated. Dashed lines indicate the beginning and end of irrigation. There was no water in B-6 after June 18.

The concentrations of both NO_3^- and Cl^- increased in B-1 beginning 28 hours after the start of irrigation, but the concentrations in the other deep wells did not. During the next site visit (June 23), concentrations in B-1 were lower than at the end of the irrigation experiment, and the concentrations in B-3 and B-4 were higher than they were during irrigation. These results could be interpreted as indicating that a "wave" of NO_3^- and Cl^- passed down to the water table and then moved outward past B-3 and B-4. This interpretation is consistent with the site-source hypothesis. Unfortunately, the deep wells could not be sampled intensively between June 20 and June 23, so the timing and intensities of any peak concentrations in the deep wells are not known. The percolation rate of 1.6 m per hour calculated during the catch basin test suggests that the concentrations of NO_3^- and Cl^- should have increased in B-1 after approximately 3.5 hours.

The highest Br^- concentration measured in the deep wells, including during irrigation experiment 5, was 0.34 mg/L. This low concentration, coupled with the absence of Br^- during the irrigation experiment, supports the off-site-source hypothesis. Bromide concentrations in T1 peaked at 12 mg/L, 31 mg/L and 143 mg/L, after the septic tank was dosed in September, March and May, respectively, corresponding to Br^- dilution factors of 35 to >400 between T1 and the deep wells. By comparison, the dilution of Cl^- between the same endpoints during the study ranged between 5- and 15-fold.

4.5.2.3 Modeling N Removal

Three modeling scenarios were explored, considering both on- and off-site septic tanks as the N source, and using B-6 or the off-site wells as the background well(s). Although there were several other OSS sources in the vicinity of the site, only the three septic tanks at the PF were available for sampling. Of those, septic tank #2 was not considered in the analysis because it was located behind the PF, much farther from the site than the PF's two other tanks. The PF's OSSs #1 and #3 are reasonable sources on which to base the modeling, because the PF generates a large volume of OSS effluent and the drainfields are at a higher elevation than the study site. Other systems surrounding the site are single-family use and are at equal or lower elevations.

To evaluate the appropriateness of the different combinations of model inputs, the TDN vs. Cl^- data for the deep well samples were graphed, along with “dilution lines” connecting various OSS sources to various background well(s). If the composition of samples from the deep wells fell near the dilution line, this outcome would be consistent with conservative transport of both NO_3^- and Cl^-. If the values fell below the dilution line, that outcome would suggest removal of TDN, which would be attributed to denitrification. Data falling above the dilution line would suggest that NO_3^- was being produced in the subsurface, which is implausible; the only reasonable explanations for such an outcome are analytical error and non-representative sampling. The three modeling scenarios are summarized in Table 4, and the dilution line graphs for each scenario are shown in Figure 45.

Table 10. Three different possible modeling scenarios.

Scenario	**Source**	**Background**
1	PF	Off-site wells
2	Site	Off-site wells
3	Site	B-6

Scenario 1 uses data from the PF’s septic tanks as sources, and data from the off-site wells as background values. Some care must be taken in evaluating the concentrations for the PF’s septic tanks, because only three samples were collected. Scenario 2 uses data from the site septic tank as the source, and data from the off-site wells as background values. Scenario 3 also uses the site septic tank data, but treats B-6 as the background well, as originally anticipated in the study design.

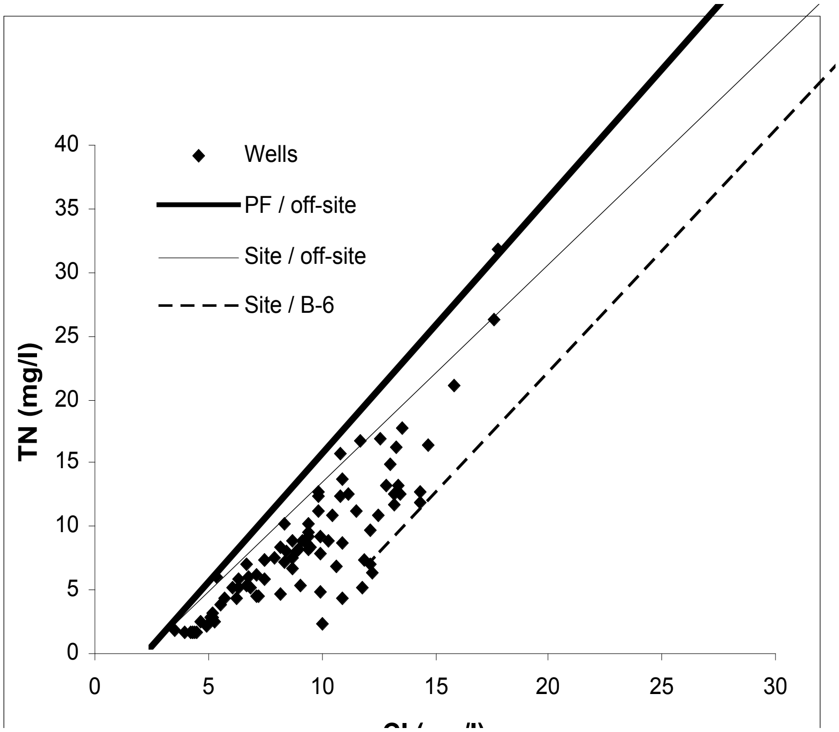


Figure 45. Deep well TDN vs. Cl^-, considering three possible dilution lines. Individual points are concentrations in deep well samples. The top bold line is a dilution line connecting the average PF sewage system values with the average off-site background wells. The thin middle line connects the average site OSS values with average off-site background wells. The dashed lower line connects the average site OSS values with average B-6 values.

Scenarios 1 and 2 appeared to be plausible ways to model TDN removal from the deep wells. Scenario 3 did not appear to represent a reasonable model because most well samples fell above the dilution line. The input values for scenarios 1 and 2, as well as the calculated median TDN removal percentages associated with the two scenarios, are listed in Table 5.

Table 11. Input concentrations (average and 1 standard deviation) and median TDN removal for 2 scenarios. Background in both scenarios is the off-site wells.

Scenario	Source	Source TDN (mg/L)	Source Cl (mg/L)	Background TDN (mg/L)	Background Cl (mg/L)	Median TDN Removal
1	PF	57 ± 9.5	35 ± 15	0.50 ± 0.26	2.5 ± 0.94	32%
2	Site	108 ± 4.6	65 ± 12	0.50 ± 0.26	2.5 ± 0.94	28%

4.5.3 Shallow Wells

4.5.3.1 General

Figures 46 and 47 display concentrations of NO_3^-, NH_4^+, Cl^- and Br^- in shallow wells T1, W2, W3 and W4 during irrigation experiments 1, 4 and 5.

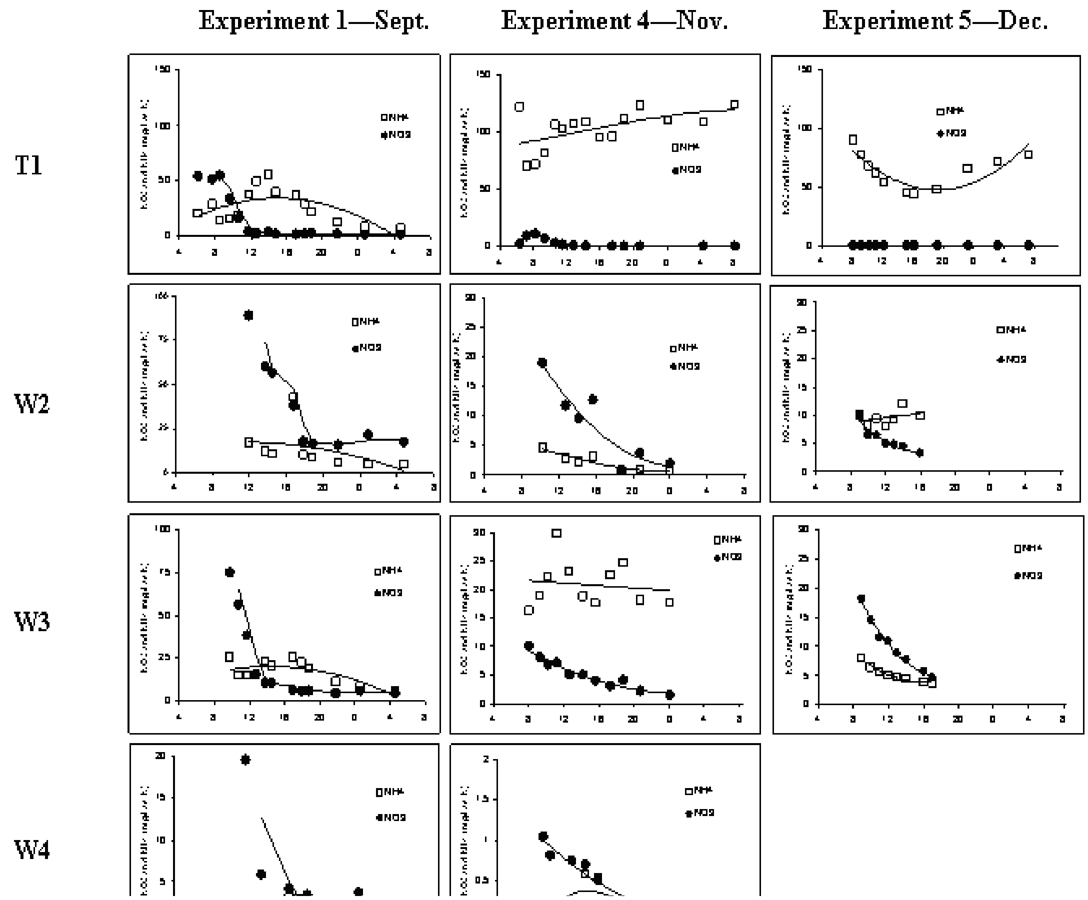


Figure 46. Concentrations of NO_3^- and NH_4^+ during the irrigation experiments. Horizontal axis is time after irrigation initiation (4 to 28 hours in all graphs). The water was on for all graphs at all times except experiment 5, when the water was turned off at 17.5 hours. Note that not all the vertical scales are the same, and that there was no water in W4 during experiment 5.

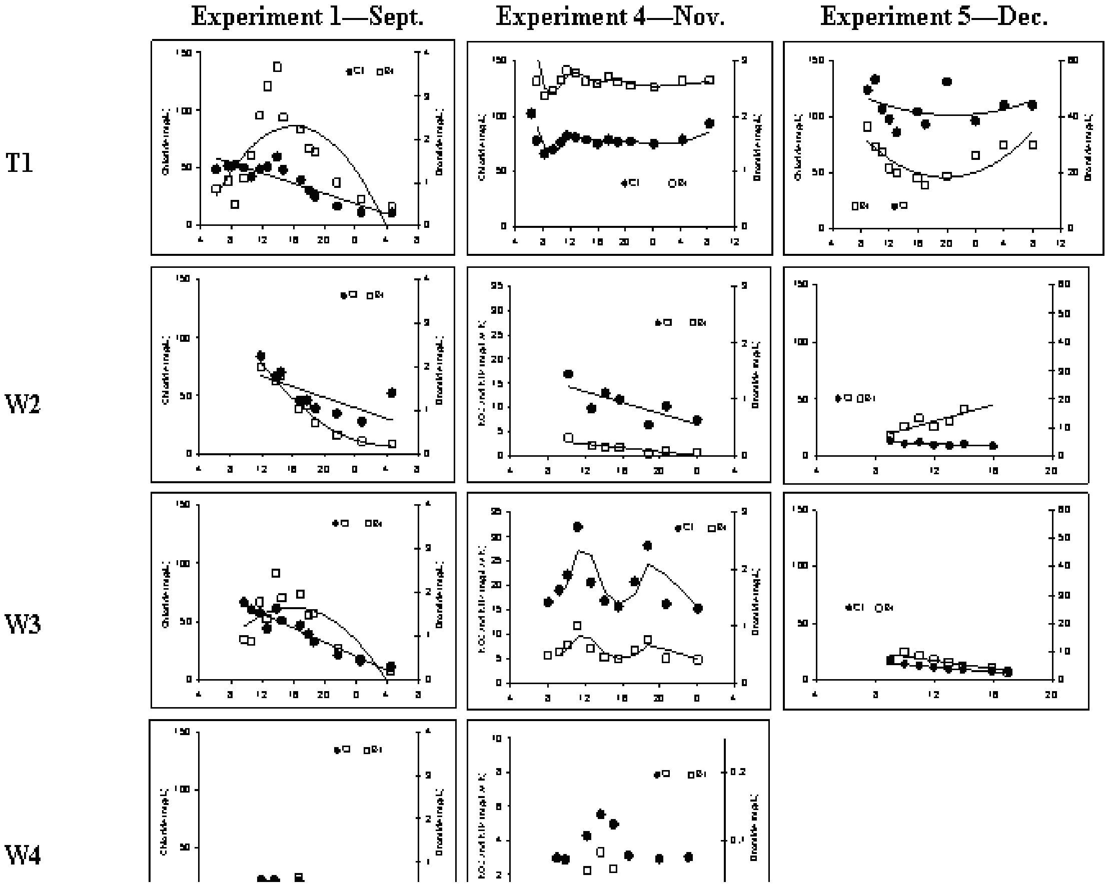


Figure 47. Chloride and Br$^-$ concentrations in shallow wells during irrigation experiments. The horizontal axis shows time after irrigation start in hours (4 hours to 24-32 hours is shown for various graphs). In experiment 5 the water was turned off at 17.5 hours. In the other experiments the water was on for the full period graphed. Note that vertical axes are not all the same, and that there was no water in W4 during experiment 5.

Concentrations of NO_3^-, NH_4^+ and Cl^- in shallow wells W2, W3 and W4 were much higher in experiment 1 than in experiments 4 and 5, a difference that might be attributed to different antecedent moisture conditions. Experiment 1 took place in September, after the drier summer months, whereas experiments 4 and 5 were in December and June, respectively, during and after the wet winter months. It is likely that experiment 1 mobilized constituents that had accumulated in the drainfield during the summer. Because Br$^-$ was added to the septic tank in increasing doses in September, March and May, the Br$^-$ concentrations were expected to increase with each experiment, which they did.

Nitrate and Cl^- concentrations started high and then decreased in almost every shallow well during each experiment. By contrast, in all shallow wells during experiment 1, NH_4^+ concentrations initially increased and then decreased. This is consistent with the mobilization-hypothesis described above for the following reasons: N in OSS effluent is almost entirely in the form of NH_4^+ as it leaves the drainpipe, and is usually completely converted to NO_3^- after traveling less than one meter (Wilhelm, 1994). Therefore, nitrogen stored in the vicinity of the shallow wells (which are thought to be located 0.5 to 1 m from the drainpipe) would be expected to be predominantly in the form of NO_3^-. Upon irrigation, NO_3^- concentrations would decrease as irrigation water mobilized and diluted the NO_3^-. However, NH_4^+ concentrations in the shallow wells would be expected to rise as the NH_4^+ rich water nearest the OSS pipe was mobilized. After some time, NH_4^+ would also be diluted because a large volume of dilution water was used during each experiment.

The shallow well Br^- data provide evidence that does not support the site-source hypothesis. Bromide and Cl^- should behave conservatively in the subsurface, as they did in most shallow wells during irrigation experiments 1, 4 and 5 (Figure 47). If the site-source hypothesis were correct, the Br:Cl ratio would be approximately the same in the deep and shallow wells. The average Br:Cl ratios of samples collected from shallow wells (excluding T1, which is nearest the drainpipe and was usually not nitrified) during the irrigation experiments were 0.024, 0.023, and 0.74. The large discrepancy between the ratio during experiment 5 and experiments 1 and 4 is due to the fact that substantially more Br was present in the system by that time. The average Br:Cl ratio of the deep wells during this study was 0.012.

4.5.3.2 Modeling N Removal

Modeling removal of N was more straightforward for the shallow wells because the OSS source was clearly the OSS at the study site. Furthermore, because a large volume of tapwater was applied during the experiments, that water could be used as the endpoint for the background concentrations. Table 6 presents the input values used to model N removal. An additional modeling step was introduced to account for the fact that not all the nitrogen was in the form of NO_3^- in the shallow well samples. Median TDN removal percentages are given in the last column of Table 6, and dilution line graphs are shown in Figure 48.

Table 12. Input concentrations (average and 1 standard deviation), and median TDN removal for irrigation experiments 1, 4 and 5.

Experiment	Source TDN (mg/L)	Source Cl (mg/L)	Background TDN (mg/L)	Background Cl (mg/L)	Median TDN Removal
1	168 ± 2.2	133 ± 12	0.32 ± 0.30	3.8 ± 0.27	65%
4	130 ± 7.1	90 ± 5.6	0.12 ± 0.01	2.9 ± 0.09	0%
5	117 ± 1.5	57 ± 0.75	0.13 ± 0.01	3.9 ± 2.1	0%

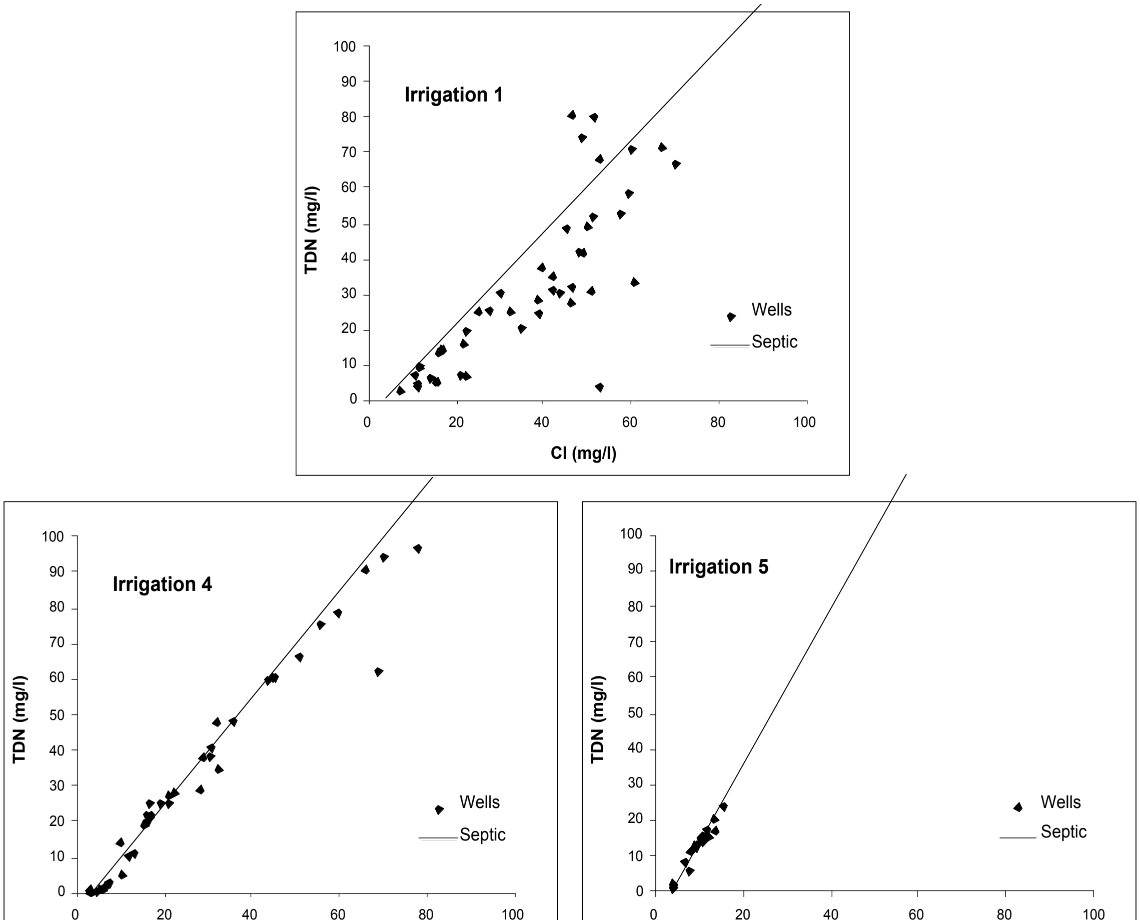


Figure 48. Irrigation 1, 4 and 5 shallow well TDN vs. Cl⁻ dilution plots.

4.5.4 Dissolved Organic Carbon and Dissolved Oxygen

Dissolved organic carbon was detected in the deep wells at low concentrations throughout the study (0.41 ± 0.22 mg/L). These concentrations yield DOC:TDN ratios of 0.04 to 0.09, which are well below the 1:1 ratio required to denitrify all nitrate in a sample. A high percentage of the DOC was removed between the septic tank and T1

(Table 3), before the NH_4 was nitrified. The DOC was therefore not available to participate in the denitrification process.

DOC was approximately constant in each shallow well, at average concentrations ranging from 2-4 mg/L in W4, 5-7 mg/L in W2, 8-10 mg/L in W3, and 14-20 mg/L in T1. The DOC:TDN ratios for each well during each experiment are shown in Table 7.

Table 13. DOC/TDN ratios for wells during irrigation experiments.

Experiment	Well	DOC/TDN
1	T1	0.3 ± 0.1
	W2	0.2 ± 0.1
	W3	0.3 ± 0.1
	W4	0.5 ± 0.1
	Ave	**0.3 ± 0.2**
4	T1	0.2 ± 0.02
	W2	0.8 ± 0.4
	W3	0.3 ± 0.02
	W4	3.4 ± 1.3
	Ave	**1.4 ± 1.6**
5	T1	0.2 ± 0.04
	W2	0.8 ± 0.2
	W3	0.7 ± 0.1
	Ave	**0.7 ± 0.2**

Dissolved oxygen remained at 8 ± 2 mg/L throughout the study in shallow and deep wells, with the exception of T1. Dissolved oxygen in T1 was 3.4 ± 1.5 mg/L.

4.6 Mason 10 Discussion

4.6.1 Site-Source vs. Off-Site-Source Hypotheses

The increases in the NO_3^- and Cl^- concentrations in B-1, B-3 and B-4 toward the end and after irrigation experiment 5 (see Figure 44) support the site-source hypothesis. Throughout most of the experiment, NO_3^- and Cl^- concentrations were relatively constant in all deep wells. At the end of the experiment concentrations in B-1 increased to their highest levels observed during the entire study. (Chloride in B-5 also changed from 11 to 15 mg/L, however a corresponding increase in NO_3^- did not occur.) The area irrigated included B-1, but not the other deep wells. During the next three site visits, concentrations in B-1 decreased to their pre-irrigation levels. During the next three site visits, B-3 and B-4 concentrations were generally lower than at the end of the irrigation experiment.

The site-source hypothesis is also supported by the fact that, throughout the study, consistent, substantial differences in concentrations existed among the deep wells. If an off-site-source located more than 50 m away was responsible for the high concentrations of NO_3^- and Cl^- in groundwater beneath the site, the differences in concentrations between the relatively closely spaced deep wells should have been relatively small.

The last piece of evidence linking the site's OSS and groundwater is the finding that the site septic tank TDN:Cl ratio increased during the study and that the TDN:Cl ratio in the groundwater also increased.

Two lines of evidence do not support the site-source hypothesis. One is the lack of a Br^- breakthrough for the deep wells during (or after) irrigation experiment 5. The other is the disparity in percolation rates observed during that experiment and the catch basin test.

The least complex explanation for the absence of Br^- in the deep wells is that the path of the OSS plume at the site has a substantial lateral component. Shallow lateral movement of OSS effluent has been documented at other sites studied in the Hood Canal watershed (Horowitz et al., 2008). For example, OSS effluent traveling west at a gradient as steep as 25^o (2.1H/1V) would pass B-5 before entering groundwater (ignoring capillary fringe and dispersion).

Given the conflicting evidence, it is difficult to positively establish the source of the majority of the plume in the groundwater beneath the site. Interestingly, the choice of whether to model either site- or off-site-source makes only a 4% difference in median estimated TDN removal, because both potential effluent sources have similar TDN:Cl ratios.

4.6.2 Nitrogen Removal

TDN removal at the site can plausibly occur only by heterotrophic denitrification. The factors that limit denitrification include oxic conditions (>2 mg/L DO) and low organic carbon availability (Korom, 1992). Low DO concentrations were never measured in the deep wells during this study, which suggests that, if denitrification occurred, it would have had to occur either before water reached the deep wells or in micro-environments within soil pore spaces and on the surface of soil grains. This possibility was not evaluated during this study.

Low DOC concentrations are thought to be responsible for limiting N removal in the groundwater beneath the site. The DOC:TDN ratios in samples collected from deep wells were always less than 0.1, which is an order of magnitude less than the ratio required for complete denitrification. Dissolved organic carbon was removed rapidly from the system: a 65% reduction was observed between the septic tank and T1, in other words, before site OSS effluent traveled far enough away from the drainfield to participate in denitrification.

Dissolved organic carbon was measured in shallow wells at higher concentrations compared to the deep wells. The average DOC:TDN ratios in samples collected from shallow wells during the irrigation experiments ranged between 0.3 and 1.4.

5. Kitsap 3

5.1 Introduction & Goals

During Year 1 of the Hood Canal Onsite Sewage System Study, five field sites were studied to assess nitrogen transformations and nitrogen removal in OSS effluents. Three of these field sites were located in Kitsap County in glacial till substrate. One site, KIT03, was particularly informative because groundwater perched after precipitation events over the glacial till layer, allowing for regular sample collection from the monitoring wells. Research at KIT03 was continued for a second year to learn more about OSS nitrogen in this system and to verify the Year 1 results.

The first year of study at the KIT03 site led to the conclusions that nitrification of the OSS effluent was nearly complete and that subsequent denitrification caused partial nitrogen removal. Denitrification requires a supply of electron donors to reduce NO_3^- to molecular nitrogen. Three possible sources of electron donors are 1) labile DOC from the OSS, 2) labile DOC stored in groundwater and/or soil, and 3) electron donors other than DOC stored in groundwater and/or soil. Oxidation of OSS DOC was nearly complete close to the drainfield. While this DOC may have contributed to the reduction of OSS nitrate, it most likely only played a minor role. It was not possible to draw conclusions regarding the supply of electron donors for the denitrification process from the Year 1 study.

The goals of the second year of study at KIT03 were:

1) to determine if the elevated chloride concentrations detected in the sampling wells during Year 1 were from the KIT03 effluent plume, and

2) to repeat sample collection and analyses to confirm the speciation of N in the subsurface and the extent of DOC and N removal in and around the drainfield.

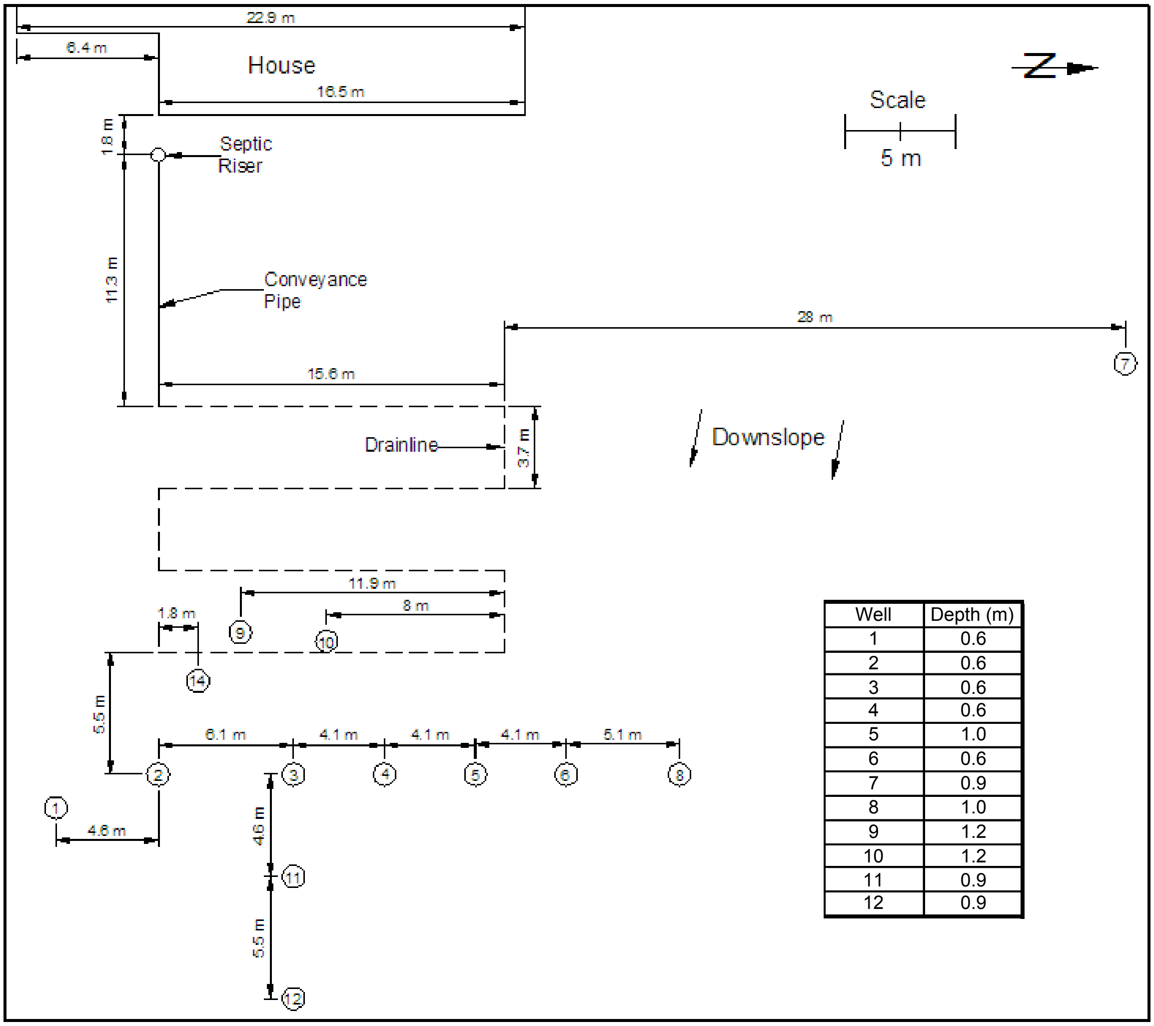
5.2 Kitsap 3 Site Description

KIT03 is a 10,000-m^2 residential site located approximately 1.6 km from Hood Canal. Geologic mapping indicates that the site lies in the Vashon glacial till, which is prevalent in the Hood Canal drainage basin (Yount, 1993). The site slopes gently to the east, and groundwater is assumed to flow in that direction. A single occupancy residence is located in the middle of the site, as shown on Fig. 49. The residence is occupied year-round by three individuals. Site surface cover includes a conifer forest around the periphery, with gravel or landscaped lawn covering the remainder of the site.

The residence is served by a conventional septic tank and drainfield. Sewage flows by gravity to a septic tank before flowing to four drainfield pipes. The drainfield is located partially in an area of landscaped lawn and partially in an area of thinned conifer forest.

During November and December 2006, 12 wells were installed to depths ranging from approximately 0.2 to 1.4 meters bgs. Well 7 is the site background well; all other wells are located in the assumed down-gradient direction from the OSS drainfield.

Three test pits were excavated with a backhoe to depths of approximately 0.9 to 1.3 m bgs on March 29, 2007. In general, subsurface geology consisted of brown, silty, fine-to-coarse sand with occasional cobbles, consistent with weathered glacial till. The soil was dense at all three test pit locations, and slow groundwater seepage was observed in test pits 1 and 2 at approximately 1.0 m and 0.23 m bgs, respectively. One well was installed in one of the test pits (Well 14). All test pits were backfilled on the same day they were excavated. An additional well (Well 15) was installed close to Well 8 in February to house a pressure transducer for measuring groundwater level fluctuations.



Well	Depth (m)
1	0.6
2	0.6
3	0.6
4	0.6
5	1.0
6	0.6
7	0.9
8	1.0
9	1.2
10	1.2
11	0.9
12	0.9

Figure 49. KIT03 site plan.

5.3 Kitsap 3 Site-Specific Methods

The general methods describing well installation and sampling are found in Section 2. Methods unique to the KIT03 study are described below.

5.3.1 Bromide Tracer Addition

Bromide was added to the septic tank as a conservative tracer. A concentrated solution of potassium bromide (approximately 5.0 g/L) was dosed from a 20-L carboy into the septic tank using a peristaltic pump at a flow rate of approximately 1.7 L/d. Tracer addition began on January 28, 2008 and ended on March 31, 2008. The carboy was refilled as needed. Based on literature values of per capita water use, this bromide dosing concentration was intended to increase the concentration of Br^{-} in the septic tank to 10 mg/L, or approximately 500 times the background concentration.

5.3.2 Background and Tap Sampling

Well 7 was the site background well, as noted in the site description section. Sample collection followed the well sampling protocol described above.

During most site visits, samples of tap water were collected in addition to the well samples. Tap water samples were collected from spigots located outside the house. Sample bottles were flushed with tap water twice before tap water was collected. The sample bottles were prepared and transported in the same way as those used for collection of well water.

5.3.3 Sampling Frequency

Between December 2007 and March 2008, 13 site visits to KIT03 were made. Table 14 shows site visitations by date, along with the total number of well, tap and OSS samples collected and analyzed.

Table 14. Field days and sample collection totals at KIT03

Dates sampled	Well samples collected	Septic tank
12/4/2007	14	1
1/24/2008	3	1
1/28/2008	3	1
1/31/2008	8	1
2/4/2008	6	1
2/7/2008	11	1
2/11/2008	9	1
2/14/2008	5	1
2/18/2008	3	1
2/24/2008	2	1
3/13/2008	2	1
3/17/2008	2	1
3/31/2008	9	0
Total: 13	**77**	**12**

5.4 Probabilistic Modeling

The probabilistic model used for KIT03 is identical to the one described for MAS10 in Section 4, however, the additional modeling step described for the shallow wells at MAS10 was not necessary at KIT03.

5.5 Kitsap 3 Results

5.5.1 Nitrogen Speciation in the OSS Plume

Figure 50 shows that NO_3^- was the dominant N species in all the sampling wells in the drainfield. Ammonium was always the dominant N species in the septic tank (mean concentration = 53.1 ± 11.3 mg/L NH_4-N), with negligible amounts of nitrate present. These results suggest that ammonium was rapidly converted to nitrate in the drainfield.

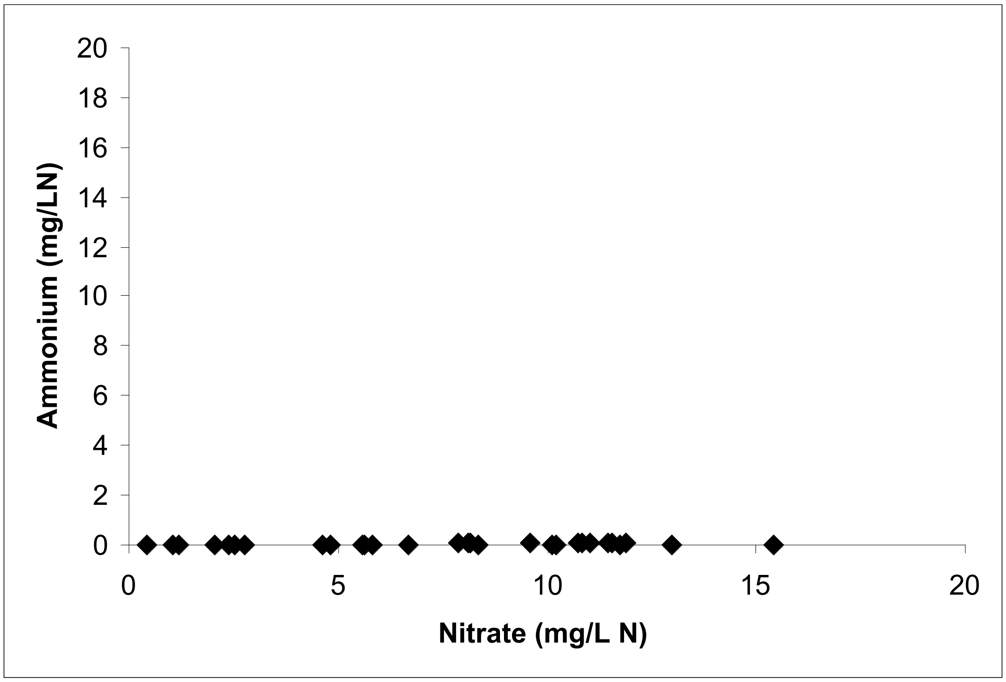


Figure 50. Nitrogen speciation in KIT03 well samples.

5.5.2 Bromide and Chloride Concentrations In and Around the Drainfield

The mean Br^- concentration in the septic tank at KIT03 was 44.0 ± 29.8 mg/L from 1/31/08 though 3/17/2008 during the period of bromide addition. Elevated Br^- in the well samples, indicating the presence of OSS effluent in those samples, was detected at Wells 3, 10, 11, 12, and 14 (Figure 51). For all the wells, samples with elevated bromide also had elevated chloride concentrations, and samples with low bromide also had low chloride concentrations.

Concentrations of Br^- in Wells 5, 6, and 8 were not significantly different from the background well (Well 7), in which the Br^- concentration averaged 0.13 ± 0.03 mg/L. Samples from these latter wells were excluded from the probabilistic modeling and subsequent analysis, because there was insufficient evidence of connection to the OSS effluent plume. Due to lack of water and therefore lack of data for Wells 2, 4, and 9, it was not possible to determine if the plume passed through these locations, so these wells were also excluded from the analysis.

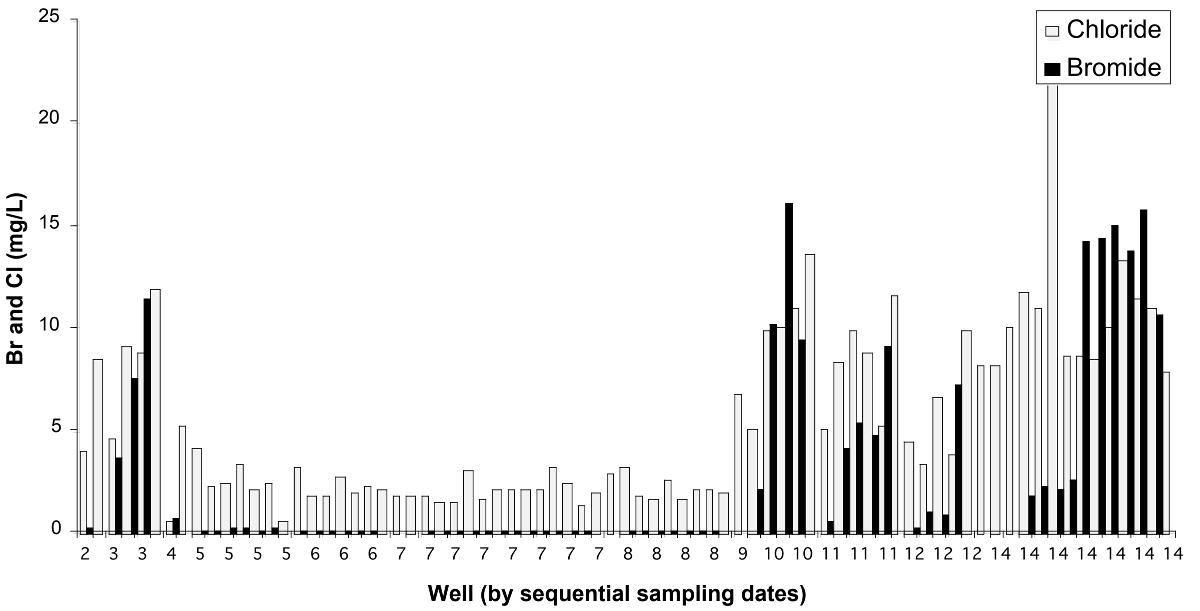


Figure 51. Chloride and bromide concentrations (mg/L) in all well samples.

5.5.3 Bromide Flowpaths

Figure 52 shows the concentration of Br^- in the wells over a period extending slightly more than the nine weeks during which time it was added to the septic tank. The concentration in Well 10, which was located directly adjacent to a drainfield line, increased rapidly to the highest concentration observed in any of the wells. The concentrations in Well 12, located farthest from the drainfield, were the lowest of any well throughout the study. The concentrations in the other three wells were intermediate between those in Wells 10 and 12.

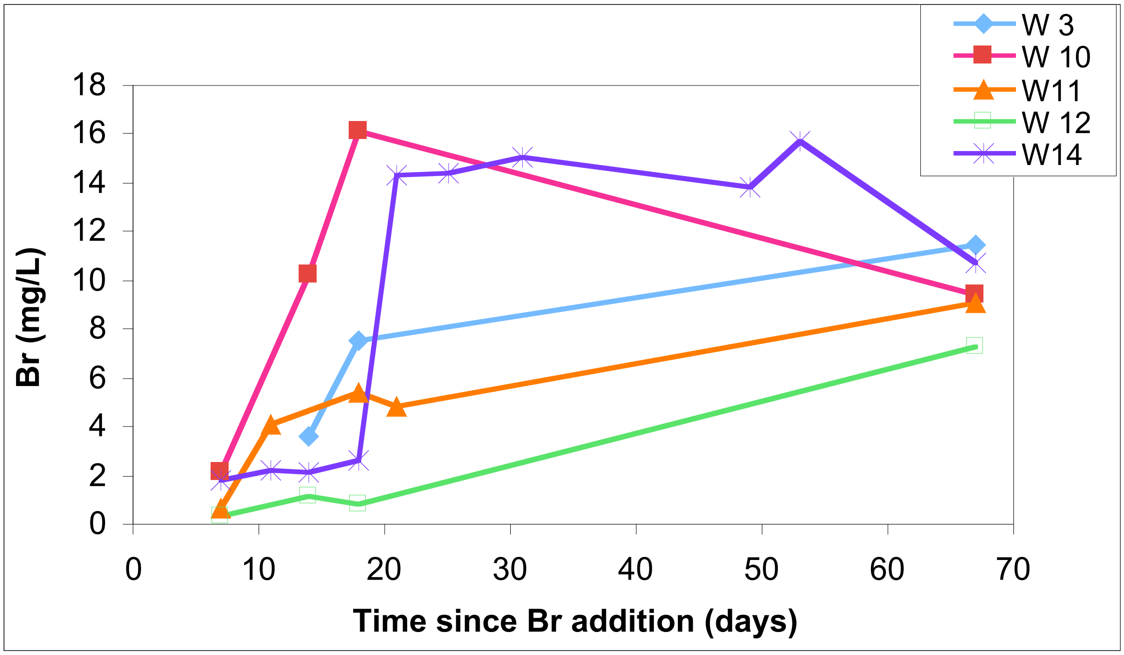


Figure 52. KIT03 bromide concentrations over time.

5.5.4 Nitrogen Removal

The median computed TDN removal in the drainfield samples, based on the probabilistic model, was 39%; the corresponding value during Year 1 was 66%. Figure 53 shows the probability of TDN removal during Year 2; the analysis suggests that most wells had partial TDN removal, some had nearly complete removal, and others had no removal.

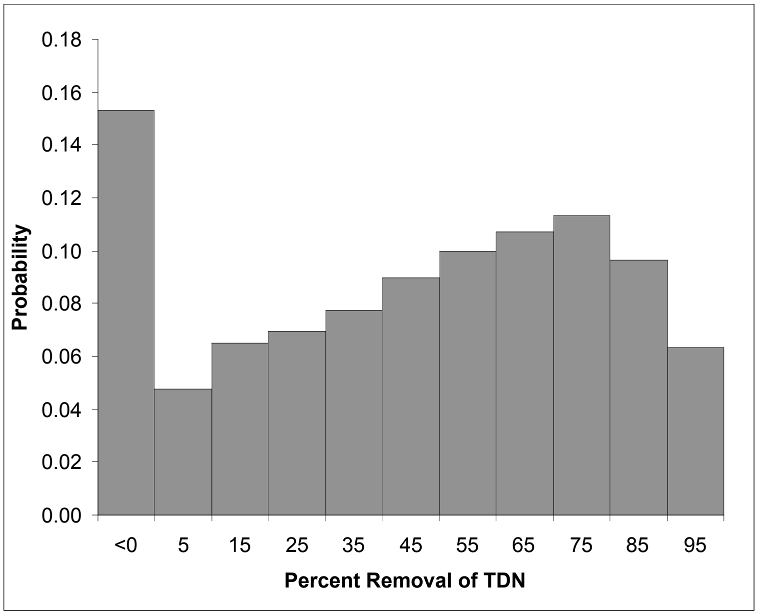


Figure 53. KIT03 Year 2 frequency distribution of estimated N removals.

Figure 54 presents the median N removal by well from Years 1 and 2. TDN removal in Wells 10 and 11 was substantially lower in Year 2 than Year 1. Removal was similar in the two years in Wells 3 and 12; Well 14 was only sampled during Year 2.

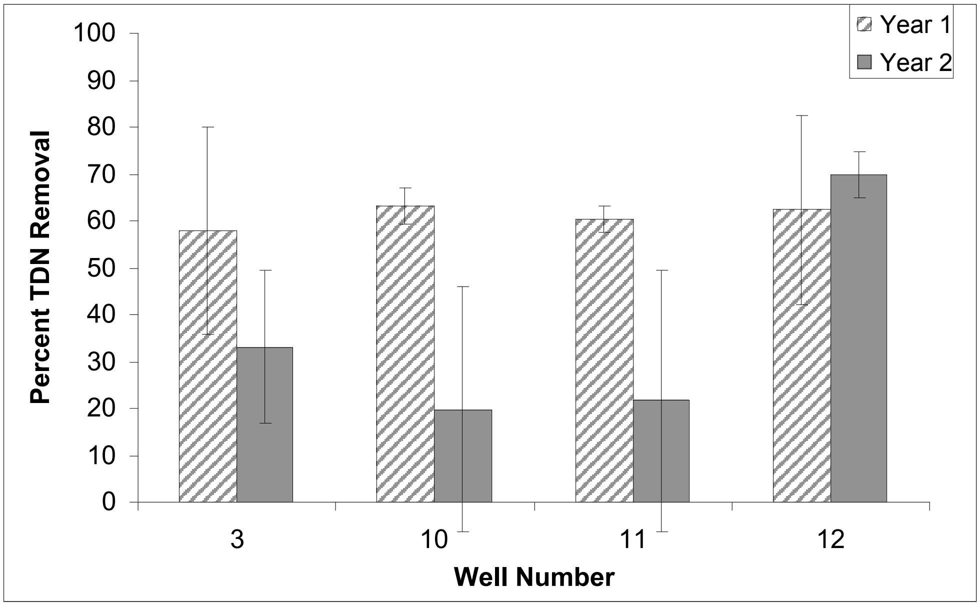


Figure 54. KIT03 median modeled TDN removal estimates by well during Years 1 and 2; error bars are one standard deviation.

5.5.5 DOC Transformations

DOC modeling results for KIT03 are presented in Figure 55. The median estimated DOC removal was 87%. The estimated removals of DOC and TDN were not closely matched; the probability of nearly complete DOC removal was substantially larger than the probability of complete TDN removal.

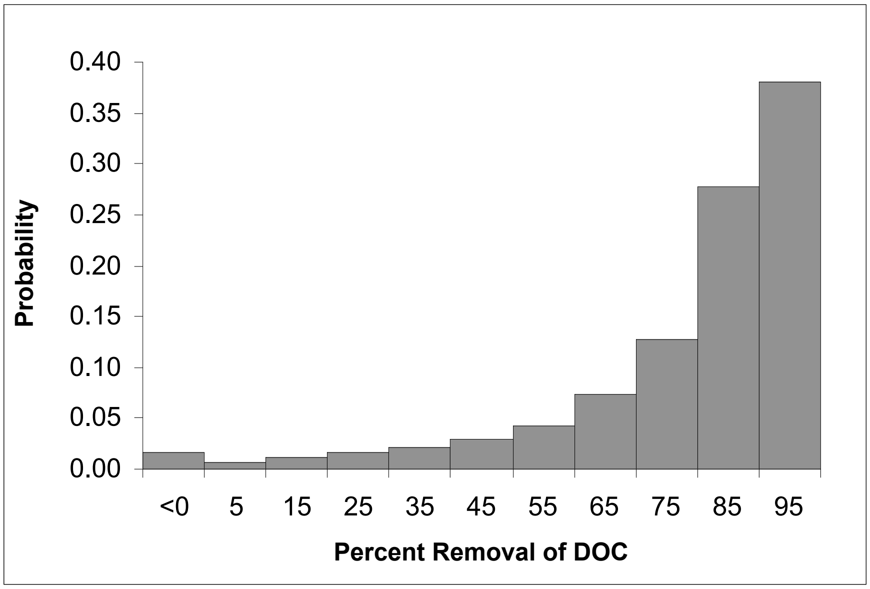


Figure 55. KIT03 Year 2 frequency distribution of estimated dissolved organic carbon.

Figure 56 presents median DOC removal by well from Years 1 and 2. DOC removal was nearly the same in all wells during Years 1 and 2.

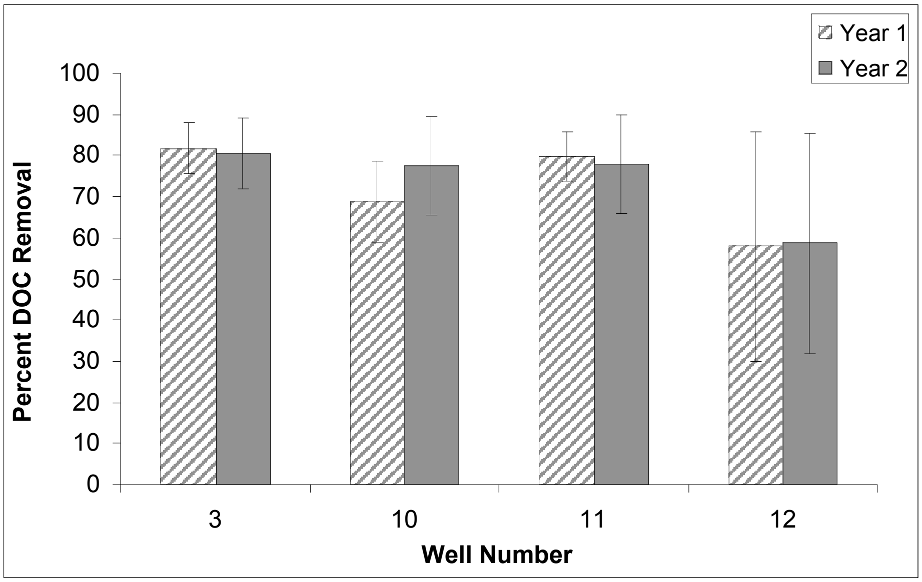


Figure 56. KIT03 median modeled DOC removal estimates by well during Years 1 and 2; error bars are one standard deviation.

5.5.6 Relationships between DOC and TDN

The mean concentration of DOC removed was consistently greater than the mean concentration of TDN removed. The median mass ratio of removed DOC to removed TDN was 4.77 ± 7.73 g C/g N. Mean measured DOC concentrations were relatively constant among all wells, while TDN concentrations decreased with decreasing chloride concentration (Figure 57). The measured mean DOC/TDN ratio in the sampling wells was 0.41 ± 0.76 g C/g N, while the mean ratio in the septic tank was 0.72 ± 0.31 g C/g N.

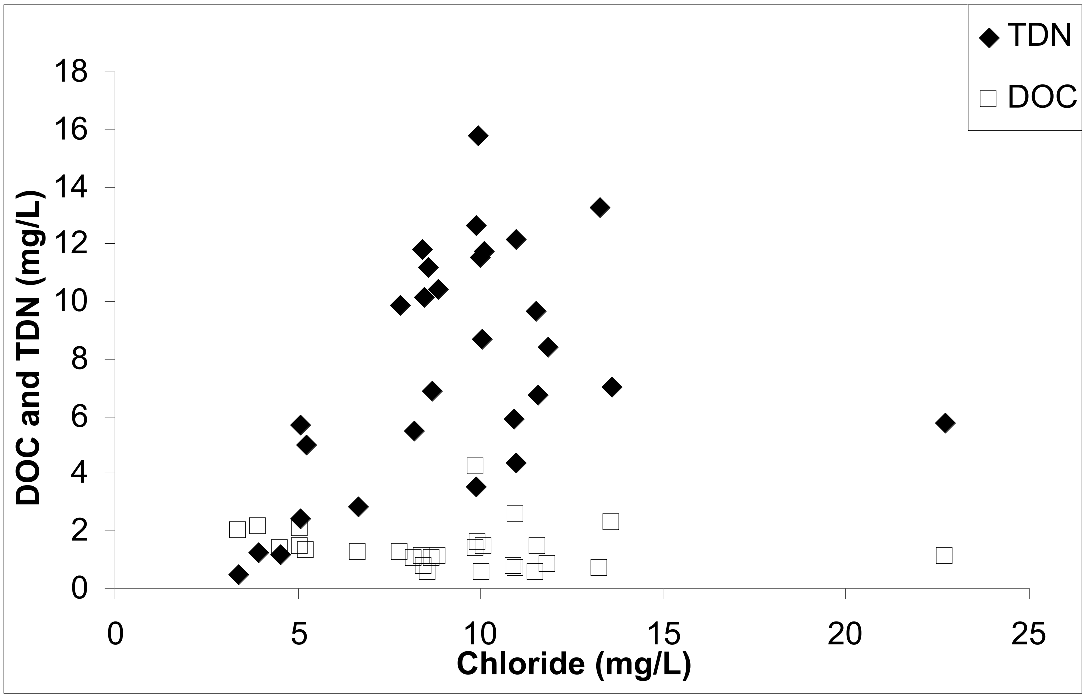


Figure 57. KIT03 measured DOC and TDN concentrations compared to chloride concentrations.

5.6 Kitsap 3 Discussion

5.6.1 Nitrogen Speciation in the OSS Plume

After leaving the septic tank, ammonium in the KIT03 OSS effluent was oxidized to nitrate. Ammonium was the dominant form of N in the septic tank, and nitrate was the dominant N species near the drainfield, indicating that the plume was exposed to oxic conditions and nitrifying microbes during subsurface transport.

5.6.2 Characterization of the OSS Plume Using Bromide and Chloride

Bromide was added to the septic tank and used to quantify the fraction of OSS effluent in the water samples. Bromide, and therefore OSS effluent, was detected in some wells and not in others (Figure 51). Samples with elevated bromide also had elevated chloride concentrations, validating the use of chloride as an OSS effluent tracer.

Bromide and chloride concentrations decreased with increasing distance from the drainfield. Since these ions are conservatively transported, decreasing concentrations can be attributed to dilution. The pattern of dilution in Wells 10, 3, 11 and 12, located progressively farther from the drainfield, suggests that the OSS plume flowed down-gradient in a westerly direction.

5.6.3 Nitrogen Removal in the OSS Plume

Partial denitrification of the KIT03 OSS effluent appears to occur close to the drainfield. The probabilistic model indicates that the median N removal at KIT03is 39% (Figure 53). Year 1 results also indicated partial N removal, although the estimated N removal was substantially higher in that year (median = 66%). Not surprisingly, N removal from some wells appeared to differ in the two years. This variability might be attributable to differences in subsurface conditions.

5.6.4 DOC Removal in the OSS Plume

The mean estimated DOC removal in the drainfield was 87% (Figure 55 and 56). The ratio of removed DOC/TDN (4.77 ± 7.73 g C/g N) greatly exceed 1 g DOC/g TDN, the approximate minimum stoichiometric ratio for denitrification. Based on this observed ratio and evidence that the OSS effluent was exposed to oxic conditions, it is likely that aerobic respiration consumed nearly all of the labile OSS DOC before it could have been utilized for denitrification.

The measured DOC concentrations in the sampling wells were statistically indistinguishable from the background DOC concentration (Figure 57), suggesting that essentially all of the OSS DOC added to the subsurface was removed, and that the DOC from the soil and groundwater was likely recalcitrant. The ratio of DOC remaining in the water that reached the wells to TDN in that water suggests that, in most samples, there is insufficient DOC available to reduce the remaining NO_3^-. Therefore, additional soil or groundwater DOC or another electron donor source would be necessary to reduce the remaining NO_3^-.

Table 15. DOC concentrations in the septic tank, sampling and background wells.

Mean Concentration	Septic Tank	Sampling Wells	Background Well
DOC (mg/L)	31.4 ±13.7	1.41 ± 0.77	1.43 ± 0.36

6. Regional Groundwater Analysis

6.1 Introduction

Shallow domestic drinking water wells were sampled 26 times during Year 2 in order to characterize the shallow groundwater that OSS effluent interacts with in the Hood Canal watershed. After percolating through the vadose zone, OSS effluent may be diluted by ambient groundwater depending upon the time of the year and the location of the OSS. The characteristics of this groundwater may determine whether or not N removal is likely to occur in the saturated zone. Specifically, the high DO and low DOC concentrations in the groundwater may could the extent of denitrification, because denitrification requires anoxic (or nearly anoxic) conditions and a source of electrons; labile DOC is the most likely source of such electrons. If data indicate that denitrification cannot proceed in the saturated zone, it can be assumed that most N contained in OSS effluent which is not removed before reaching the water table will be transported conservatively to Hood Canal.

6.2 Methods

A database of domestic drinking water wells was compiled using DOE's public records. The criteria for well selection were that the well was located in the Hood Canal watershed and that it was relatively shallow, with a depth less than 60 feet. Well owners were contacted by mail and then by phone and were assured that the data collected in the study could not be traced to specific wells. Twenty-six homeowners granted access to their wells, and single samples were collected from each well. Samples were collected as close to the well as possible, i.e., before the water had passed through a holding tank. If a sampling port upstream of the holding tank was not available, samples were collected from a holding tank port or faucet. In all instances, water was allowed to flow freely for thirty seconds prior to sample collection. Samples were collected into acid-washed, labeled 250-ml HDPE bottles and stored in a cooler on ice for transport to the laboratory. Following sample collection, an additional 40 mL of water was collected in a clean glass beaker for immediate determination of DO, pH, conductivity and temperature. Drinking water well samples were filtered in the lab and analyzed for DOC, TDN, NO_3, and NH_4.

6.3 Results

Constituent concentrations measured in the well samples are shown in Table 16. DO concentrations were generally high, with an average of 6.5 ± 2.0 mg/L. All samples contained at least 0.2 mg/L DO, the threshold for oxic conditions. DOC concentrations were less than 0.35 mg/L in all but one of the 26 samples. The average TDN measured in samples was 0.51 ± 0.78 mg/L; the high standard deviation relative to the mean indicates that the data is skewed. The effect of the skewed data set can be minimized by using the median TDN value of 0.23 mg/L to characterize the samples. The TDN exceeded 1 mg/L in only three samples (WS-3, WS-5, WS-10). When TDN was detectable, nitrate accounted for the all of the N. Ammonium was not detected in any drinking water well samples.

Table 16. Drinking water well results.

Sample ID	DO (mg/L)	DOC (mg/L)	TDN (mg/L)	NO$_3$ (mg/L)	NH$_4$ (mg/L)
WS-1	7.33	ND	0.06	ND	ND
WS-2	2.07	12.27	0.06	ND	ND
WS-3	4.05	0.08	1.08	1.05	ND
WS-4	5.44	0.13	0.29	0.23	ND
WS-5	6.86	0.15	3.84	3.63	ND
WS-6	6.20	0.18	0.16	ND	ND
WS-7	8.91	0.08	0.28	0.18	ND
WS-8	6.65	0.20	0.13	ND	ND
WS-9	7.38	0.33	0.16	ND	ND
WS-10	6.84	0.17	1.66	1.60	ND
WS-11	7.11	ND	0.15	ND	ND
WS-12	8.99	0.09	0.50	0.47	ND
WS-13	8.97	0.09	0.25	0.20	ND
WS-14	5.08	ND	0.12	ND	ND
WS-15	3.61	0.06	0.76	0.74	ND
WS-16	5.71	0.07	0.52	0.52	ND
WS-17	5.87	ND	0.51	0.49	ND
WS-18	5.30	0.20	0.52	0.48	ND
WS-19	6.40	ND	0.08	ND	ND
WS-20	8.54	0.10	0.20	0.13	ND
WS-21	6.84	0.12	0.16	0.09	ND
WS-22	7.26	0.12	0.93	0.94	ND
WS-23	2.38	0.13	0.16	ND	ND
WS-24	9.36	0.25	0.28	0.23	ND
WS-25	9.44	ND	0.11	ND	ND
WS-26	5.27	0.22	0.21	0.13	ND
Average	6.46	0.59	0.51	0.44	-
STDEV	2.00	2.38	0.78	0.76	-
Median	6.75	0.11	0.23	0.16	-
Min.	2.07	0.03	0.06	0.05	-
Max.	9.44	12.27	3.84	3.63	-

Detection Limits:

DOC = 0.05 mg/L, TDN = 0.05 mg/L, NO$_3$ = 0.09 mg/L and NH$_4$ = 0.15 mg/L

6.4 Discussion

This survey suggests that N removal from OSS effluent would be unlikely to occur once effluent reached the groundwater table as represented by the shallow drinking water well samples collected. The saturated zone within the Hood Canal watershed would typically not provide a suitable environment for denitrification due to relatively high DO and low DOC concentrations. Average DO concentrations are well above the 0.2 mg/L threshold for anoxic conditions, and average DOC concentrations are not adequate for complete denitrification to occur. Although such conditions are predominant in the saturated zone, denitrification may occur within micro-sites in the heterogeneous subsurface. The samples collected from WS-2 and WS-23 likely represent localized conditions existing in the subsurface that could potentially permit denitrification. For example, if small anoxic zones were present in the saturated zone near WS-2, the substantial supply of DOC could potentially be utilized for denitrification.

7. Overall Discussion

The Hood Canal Onsite Sewage System Nitrogen Study characterized N removal from effluent plumes emitted by eight OSSs in the Hood Canal watershed. The second year of this study addressed the challenges identified during the first year. These challenges were caused by the varied conditions existing in the Hood Canal watershed, which required unique methods to assess N removal at different field sites, in addition to limited monitoring event capabilities and variable seasonal conditions. The variability observed in experimental results was attributed to the distinct characteristics of each study site. This study found that there is no obvious typical OSS environment or scenario that can be generalized to the Hood Canal watershed. However, the case studies in this report and the Year 1 report (Horowitz et al. 2007) describe different scenarios that can inform our understanding of similarities and differences influencing N removal from OSS effluent within the watershed. The purpose of this section is to evaluate these separate sub-studies to obtain a broader assessment of the fate of N from OSS in the subsurface surrounding drainfields.

Table 17 summarizes the site and experimental information, and also presents the median estimated N removal values at each site. The first row of the table presents experimental information for an "ideal" site. An ideal site would be one in which the source of OSS effluent was well known, the effluent flow path was well characterized (by more than one tracer), and for which reliable background constituent concentrations were known. Additionally, the ideal site would be one from which many samples could be collected over two years of the study. The Percent Sampling column of the table compares the number of samples collected with the targeted number of "10 samples per sampling point" specified in the project contract. The value accounts for the number of well samples collected and used in estimating N removal. The value was calculated by dividing the number of samples collected by the number of wells at the site (excluding wells that never yielded any water) and multiplying by 10 to convert to a percentage.

As can be seen from the table, a high confidence can be placed in the median N removal estimates from KIT03. Moderate confidence can be placed in the estimates for the

MAS02, MAS05 and MAS10 sites. Due to limited groundwater, confidence in the estimates for MAS01, KIT01 and KIT02 is low.

Table 17. Site and experimental information.

Site Information			Median N		Experimental Information				
Site	**Location**	**OSS Type**	**Y1**	**Y2**	**N Source**	**Tracer**	**Background Source**	**Percent Sampling** [3]	**Time Period**
Idealized	*NA*	*NA*	*NA*	*NA*	*OSS*	*Cl^- and Br^-*	*Ambient Groundwater/Seawater*	*100*	*2 Years*
MAS01	Intermediate	Pump[1]	15%	NA	OSS	Cl^-	Tap Water	59	1 Year
MAS02	Intermediate	Mound[2]	92%	NA	OSS	Cl^-	Ambient Groundwater	60	1 Year
MAS05	Shoreline	Gravity	NA	7%	OSS	Br^- Reverse	Seawater	60	1Year
MAS07	Shoreline	Gravity	NA	12%	OSS	Br^- Reverse	Seawater	27	1 Year
MAS10	Intermediate	Gravity	NA	37%	Unsure	Cl^-	Off-site Groundwater	150	1 Year
KIT01	Inland	Gravity	0%	NA	OSS	Cl^-	Tap Water	16	1 Year
KIT02	Intermediate	Gravity	81%	NA	OSS	Cl^-	Tap Water	7	1 Year
KIT03	Inland	Gravity	66%	39%	OSS	Cl^- and Br^-	Ambient Groundwater	88	2 Years

[1] Septic tank pumped to drainfield, activated by float switch
[2] Septic tank pumped to elevated drainfield
[3] Number of samples collected as a percentage of project target

The key components in evaluating the fate of OSS N are: 1) identifying and quantifying the source of N, 2) finding and sampling the OSS flow path, 3) measuring concentrations of N and a conservative tracer along the OSS flow path, 4) determining the concentrations of N and a conservative tracer in the primary background source, and 5) modeling N removal from the OSS effluent.

The connection between the septic tank and the OSS flow path was identified using conservative tracers. The OSS effluent signals at MAS05 and MAS07 were quantified using a mixing-model that employed Br^- as reverse a tracer. The OSS flow path was not clearly identified at MAS10 due to the high density of OSSs near the site. The OSS plume was identified at KIT03 and MAS02 using Cl^- as a tracer at both sites and additionally by Br^- at KIT03. During Year 1, limited groundwater availability at MAS01, KIT01 and KIT02 lowered the confidence that a significant portion of the OSS plume was identified; however the limited results are still informative of N removal.

The OSS effluent flow paths had distinct characteristics at each site and required specific sampling approaches. The OSS effluent flow paths at MAS05 and MAS07 were very short and shallow due to their proximity to Hood Canal. These plumes were sampled by shallow wells located within and down-gradient of drainfields. During Year 1 several sites were excluded from the study because water was not accessible at a shallow depth.

This lack of shallow water also posed a problem at MAS10, so two methods were used to sample the presumed effluent plume: deep wells and artificial rainfall experiments. At KIT03, a shallow effluent flow path in perched groundwater was accessible using shallow sampling wells during the wet season. Our initial objective at all field sites was to collect well samples as far away as 100 ft from the drainfields (i.e., the OSS distance to the surface water compliance point) but in almost cases OSS effluent was diluted to levels indistinguishable from background conservative tracer concentrations at that distance.

Finding consistent, similar background sources was possible at four study sites. Specifically, seawater was used as a background source at the MAS05 and MAS07 sites and ambient groundwater unaffected by OSSs was used at MAS02 and KIT03. Groundwater was also used at MAS10 but the planned background well was not indicative of background conditions (due to high Cl^- concentrations), so nearby off-site wells were used. During Year 1, tap water was used as the background source at MAS01, KIT01 and KIT02 due to the inability to collect ambient groundwater.

Concentrations of N and conservative tracers were measured in samples collected from wells, septic tanks, and background sources. The number of samples collected at each site affected the level of confidence characterizing the results and conclusions of each case-study. The experimental protocol proposed that 10-12 samples were to be collected from each sampling point. The average number of samples collected from usable sampling points was equal to or greater than this at MAS02, MAS05, MAS10, and KIT03. However, the inability to consistently collect subsurface water using shallow wells at MAS01, MAS07, KIT01 and KIT02 lowered the confidence level characterizing the results of those case-studies.

Probabilistic modeling was used to estimate the extent of N removal from OSS effluent along subsurface flow paths near the drainfield. At MAS05 and MAS07, little to no N removal was found. The fairly conservative transport of N was attributed to the short hydraulic residence time of the effluent plumes in the subsurface. The partial N removal found at MAS10 and KIT03 suggests the effluent interacts with anoxic conditions and electron donors at some point, but not enough to permit complete denitrification. In contrast, the MAS02 effluent, studied during Year 1, was almost completely denitrified likely due to transport through organic rich soils, which have more available electron donors. The N removal levels calculated during Year 1 for MAS01, KIT01 and KIT02 differ greatly but the results for these sites are similar in the fact that they are characterized by a low degree of confidence due to the small sample sizes.

In conclusion, this study found that the diverse subsurface environments within the Hood Canal watershed result in highly variable N removal from OSS effluents. This field study of denitrification supports the conclusions of previous field studies that the occurrence and extent of denitrification is spatially specific and highly variable. An analysis of literature denitrification rates found that in most cases there is little to no removal and in few cases there is rapid, extensive removal (James et al, unpublished). In contrast, this study found partial N removal in several cases, complete removal in one case, and little to no removal in two cases.

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