Quantifying Nitrogen Removal by Innovative Alternative Septic Systems and Potential for Enhanced Nitrogen Removal by Labile Carbon Addition

FINAL REPORT

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PREPARED BY:

Christopher Neill Woods Hole Research Center 149 Woods Hole Road Falmouth, MA 02540

Background

Nitrogen (N) pollution is the number one water quality problem in Buzzards Bay and its large number of estuarine harbors and embayments. Excess N that enters estuaries from watersheds fuels a cascade of changes including greater algae growth, loss of eelgrass habitat, periodic low oxygen, fish kills and reduction of fish and shellfish. This is a critical regional environmental and economic issue for both Buzzards Bay and the broader coastal U.S. Many of the estuarine waters of Buzzards Bay currently do not meet federal Clean Water Act standards and the U.S. Environmental Protection Agency (EPA) now classifies water quality in the majority of estuaries in the Northeast U.S. as "fair" (EPA. 2012)

The number one N source to Buzzards Bay is wastewater, which is treated either by collection in centralized wastewater facilities, or by on-site septic systems. Currently 52% of the parcels in the Buzzards Bay watershed use septic systems (Buzzards Bay National Estuary Program, 2015) and the vast majority of these are standard "Title 5" systems that meet current municipal regulations but remove relatively little N (Costa et al. 2002). Managing N releases from complex landscapes will require a variety of approaches that range from centralized wastewater treatment to more decentralized approaches (Schipper and Davidson 2010). Because much of the area of the Buzzards Bay watershed will remain below population densities that make centralized wastewater collection and treatment economically favorable, improvement of N removal by septic systems will be critical for reducing N pollution to Buzzards Bay.

Innovative alternative (IA) systems have great potential to improve N removal by on-site septic systems. IA systems create conditions that favor denitrification—the process that converts nitrate to harmless di-nitrogen gas. However, the performance of IA systems installed in residences in Massachusetts is highly variable and ranges from more than 90% to no better than Title 5 systems (Heufelder et al. 2007). In these cases, removal efficiency was determined based on typical wastewater concentrations and removal that occurs in the septic system itself. These measurements are important to evaluate the actual amount of N that reaches groundwater and estuaries. In addition, because easily-utilizable (labile) carbon (C) is needed for denitrification, and because concentrations of labile C are often low in the output of IA systems and in the region's soils and groundwaters into which effluent is released (Pabich et al. 2001), labile C additions at the point of effluent release from IA systems into leaching fields have the potential to substantially increase N removal (MacQuarrie et al. 2001, Schipper et al. 2010).

Methods

Location of project

The project took place in the West Falmouth Harbor (WFH) watershed. WFH exhibits symptoms of N pollution including eelgrass loss, reduced benthic diversity and periods of low oxygen in the inner harbor. Significant water quality degradation in WFH, caused in part by the opening of a Falmouth wastewater treatment facility in the WFH watershed in 1986, led to WFH being one of the first Buzzards Bay estuaries for which a Total Maximum Daily Load (TMDL) for N was developed. The Town of Falmouth upgraded its wastewater treatment facility to tertiary treatment in 2005 and significant reductions in the amount of N from the

plant reaching the WFH shoreline are expected, leaving septic systems as the largest N source to WFH.

We installed groundwater wells downgradient of two newly-installed IA septic systems and one newly-installed Title 5 system at three private homes near WFH (Table 1). We used the wellfields to delineate the groundwater plumes that emanated from the septic systems. The IA systems were installed as part of the *West Falmouth Harbor Shoreline Septic Remediation Project*, which provided incentives to homeowners to upgrade existing Title 5 septic systems to denitrifying IA systems.

Home address	Septic system type
42 Chase Road	IA (Hoot)
140 Old Dock Road	IA (Hoot)
Nahsawena Road	Title 5

Table 1. Study site locations and types of septic systems studied.

Installation of well fields

We instrumented the leaching field of each septic system with three soil solution samplers (tension lysimeters) at depths of 1 m and 3 m. The tension lysimeters (Soil Moisture Equipment Corp., Santa Barbara, CA) allowed sampling of water from the soil unsaturated (vadose) zone. Each lysimeter is a PVC plastic tube connected to a semi-porous ceramic cup. The tube is sealed at the top, and when a vacuum is applied, water moves from the soil through the ceramic cup and into the tube, where it can be collected.

We also instrumented each leaching field with multiple groundwater sampling wells (Fig. 1-3). Each well had multiple depths. One depth was near the surface of the groundwater and the others deeper into the groundwater. These wells consisted of 3.8 cm (1.5 in) stainless steel drive points (AMS, Inc.) with a screen and 0.64 cm (0.25 in) outside diameter nylon tubing that extended to the surface. The micro-wells were inserted with a 1.6 cm (0.625 in) diameter pipe that was driven to about 60 cm below the water table and then withdrawn, leaving the points and tubing in place. They were installed with an electric hand-held drive hammer. Access to these wells was through narrow plastic tubing that emerged at the soil surface protected by a PVC ring. Groundwater from the wells was sampled after flushing by drawing into plastic syringes.

We surveyed the elevation of the top of the well casings, and used those elevations and measured depth to groundwater in each well. We then measured the depth to the groundwater in each well to create a topographic map of the surface of the groundwater at each location. The exact arrangement of the wells and well depths depended on topography, groundwater elevations and yard features.

Several factors resulted in the project focusing on tracking effluent from the IA system at 42 Chase Road.

First, the groundwater contours at 67 Nashawena Street were extremely flat over the property on which we had access for well installation, and in the area between the leaching

field and West Falmouth Harbor (Fig. 4). This made it almost impossible to track an actual plume from the leaching field.

Second, we had problems with regular access to the property at 140 Old Dock because the house was occupied by renters for several months.

In contrast, the property at 42 Chase Road had a gradient of groundwater elevations that was ideal for identifying and tracing a groundwater plume (Fig. 5). We therefore invested most of our experimental efforts at plume tracing at this site and installed 19 multi-depth wells to define and sample the plume from the installed IA system.

Ambient groundwater sampling and analysis

We sampled ambient groundwater from the wells at 42 Chase Road on three dates. We extracted water with 60 mL syringe, filtered it through a GF/F filters in the field into a 20 mL scint vial. Samples were brought to the laboratory on ice where they were analyzed by cadmium reduction on a Lachat 260 series autoanalyzer. Ammonium was measured colorimetrically by the phenol-hypochlorite method. Methodology followed an established quality assurance protocol that is in place for the Buzzards Bay Coalition's Baywatchers nutrient monitoring program (Williams and Neill 2014).

Plume tracer experiments at 42 Chase Road

Injection 1: Determination of timing of water flow

We simulated the plume emanating from the IA septic system by injecting NO_3^- into the groundwater in our constructed wellfield. We pumped up 36 liters of groundwater from the injection well near the leaching field (Fig. 6), to which we added a pre-mixed solution of 500 μ M NaNO₃, 20 mL rhodamine, and 1 L deionized water (DI) to the groundwater and mixed thoroughly. The water was pumped back down into Well 1s using a field peristaltic Geopump (Geotech Environmental Equipment). We took samples from each well in Fig. 6 over 95 hours. We set time 0 for this experiment as 15 minutes after all the water was pumped back down. Injecting a large volume of water (37 L) into a tightly porous reservoir created a bulge of liquid in the groundwater, so some water flowed backward against the normal flow direction. By waiting 15 minutes, this bulge has equilibrated and water again flowed in the normal direction.

We collected a total of 195 samples over 95 hours. The rhodamine dye acted as a tracer. It has a deep red color, and diluted as it moved away from the injection sites, so as we pulled samples we could check for color and absorbance to determine if the injected fluid had reached a downstream well.

We analyzed for nitrate and rhodamine (Rh) concentration in water samples. Pigmented nitrate samples were diluted before Lachat analysis and then run with and without the cadmium column to determine the color interference from Rh. Rhodamine was measured by fluorescence on a Spectra Max M2 Microplate Reader. Light was emitted at 530 nm and measured at 555 nm.

Injection 2: Nitrate addition and NO₃ attenuation

We pumped up 36 L of groundwater from the injection well, to which we added a premixed 1000 μ M NO₃ and 1000 μ M KBr solution. Sampling methods were the same as injection 1. Some samples were taken using a Geopump to pump water into a 60 mL biological oxygen demand (BOD) bottle. The second injection lasted 48 hours and included 12 total sampling times. We collected 77 total samples. Ammonium samples were preserved with 10 μ L of 6M HCl and analyzed colorimetrically. We measured the conservative tracer, bromide, on a Dionex ion gas chromatograph. The amount of NO₃⁻ or Br⁻ present was calculated as a percent remaining of the original injected concentration. Nitrate attenuation was calculated as the difference between the expected and actual nitrate concentration. Expected NO₃ was determined by calculating how much bromide had been diluted by groundwater.

Injection 3: Carbon addition and enhanced denitrification

Using a Geopump, we pumped up 36 L of groundwater from the injection well, to which we added a solution containing 1000 μ M NO₃, 1000 μ M KBr, and 0.1%v/v ethanol. Ethanol was chosen as a carbon source because of its low cost, its availability to a diverse microbial community (Cherchi et al, 2009), and the high rates of denitrification from wastewater that it can stimulate (EPA, 2013). We injected 0.9 g Chemical Oxygen Demand (COD)/L H₂O (assuming that ethanol contains 907g COD/L, which resulted in 1 mL ethanol/L H₂O). We filled BOD bottles to determine dissolved oxygen (DO) content. We preserved NH₄⁺ samples with 5N HCl. Nitrate, ammonium, bromide, and oxygen were determined using methods described above. The third injection lasted 43 hours and included 12 sampling intervals. We collected 55 total samples.

Because attenuation of NO_3^- in groundwater can be attributed to denitrification and because we were interested to see if low-oxygen conditions suitable for denitrification existed in the groundwater, we measured oxygen concentration at selected time points in the NO_3^- and $NO_3^$ plus carbon additions using a Hach LTR oxygen probe.

Results and Discussion

Groundwater contours at 42 Chase Road

Groundwater elevations showed a regular pattern of movement from east to west and steady but fairly uniform elevation drop across the 42 Chase Road property (Fig. 7). This indicated that any plume emanating from the leaching field should move to the west and be detectable in downgradient wells.

Vadose zone and ambient groundwater nitrate concentrations

The mean concentration of NO₃⁻ collected from tension lysimeters installed under the leaching field of the IA system was 1,266 μ M at 1 m and 621 μ M at 2.5 m. Mean vadose zone concentration in the yard not under the leaching field was 161 μ M. The concentration in the shallow lysimeters (equivalent to 17.7 mg N/L) was in the range of effluent concentrations from IA systems (Huefelder et al. 2007). The decrease in concentration between the shallow and deep lysimeters indicated attenuation in the vadose zone of approximately 50 percent between soil depths of 1 and 2.5 meters. In one set of lysimeters the NO₃⁻ concentration decreased from 2000 μ M at 1 m to 550 μ M at 2.3 m, a decrease of about 1000 μ M per meter of vadose zone (Fig. 8).

Groundwater NO_3^- concentrations at a depth of 0.5 m into the groundwater ranged from 60 to 200 μ M and increased rather than decreased between the leaching field and the edge of the West Falmouth Harbor estuary (Fig. 9). We interpreted this to indicate that groundwater from upgradient in the watershed was emerging near the coastline. The pattern at a depth of 1.5 m

into the groundwater was in a similar range of 80 to 160 μ M but was spatially more complex, with high concentrations both upgradient and downgradient of the leaching field (Fig. 10). This pattern also indicated complex groundwater movement with likely nitrate sources upgradient in the watershed.

In the three wells used for the tracer injection, NO_3^- increased with depth and ranged from about 50 µM at 0.5 meters to 250 µM at 1.5 meters below water table (Fig. 11). Ammonium concentrations were very low and were insignificant compared with NO_3^- (Fig. 11). This indicated that these wells were suitable for the tracer experiment because they did not differ substantially in their ambient concentrations.

Tracer experiments

Injection 1:

The rhodamine dye tracer appeared sequentially in wells 1, 2 and 3 (Fig. 12). The rhodamine concentration indicated that the bulk of the water mass passed through Well 1 in twelve hours, arrived at Well 2 after 20 hours, and passed Well 3 after 32 hours. The concentrations were most uniform in the wells 1.5 m below the water table (shown) and we used these wells for interpreting subsequent tests. At Well 1, the maximum amount of nitrate was 100% of original, in Well 2 NO₃⁻ peaked at 61 percent of injected concentration, and in Well 3 it peaked at 54% of the injected concentration. About 50 percent of water leaving the leachfield stayed fairly concentrated in the dominant direction of groundwater flow, and the remaining water dispersed with mixing.

The timing of the rhodamine passage was delayed compared with NO₃⁻, almost certainly because the rhodamine sorbed to soil particles in the aquifer. The rhodamine was an excellent visual method for quick detection of plume direction and timing, but because of the soil sorbtion, we used KBr for subsequent tracer tests.

Injection 2: KBr and NO₃⁻ addition

Nitrate concentrations closely followed the Br⁻ tracer (Fig. 13). Bromide cannot be taken up biologically and we attributed its decrease in downgradient wells to dilution. Nitrate did not decrease compared with the tracer, indicating that downgradient declines in NO₃⁻ concentrations were caused by dilution and not attenuation. Wells 2 and 3 captured the most direct plume of groundwater from the injection. In each well, the peak of Br- NO₃⁻ concentrations represented the passage of the groundwater plume. Each subsequent peak was lower because water also diffused into the plume from surrounding groundwater.

The injected mixture was 990 μ M NO₃⁻ and 968 μ M Br⁻. The initial concentration of Well 1 was 116 μ M Br⁻ and 45 μ M NO₃⁻. At 0.25 hours post-injection, concentrations peaked at almost 100 percent of their initial value.

In Well 2, the initial NO_3^- concentration was 70 μ M. The tracer began entering the well 4 hours post-injection, concentration peaked at 14 hours, and declined to background after 20 hours. At their maximum, Br⁻ and NO_3^- were 68 and 69 percent of their original concentrations. Nitrate peaks were nearly identical to those of Br⁻, indicating that most NO_3^- was not attenuated.

Well 3 intercepted the tracer downstream of Well 2. The initial NO₃⁻ concentration of Well 3 was 51 μ M. The tracer was detected 20 hours after, its concentration peaked 27 hours, and it

returned to background by 37 hours. At its maximum, NO_3^- and Br^- were 47 and 53 percent of their original concentration. We calculated no NO_3^- attenuation between Wells 2 and 3. In all wells, NH_4^+ made up less than 1% of the total dissolved inorganic N.

Injection 3: Carbon addition

Attenuation of NO_3^- did not increase with carbon addition (Fig. 14). The patterns of change in tracer and NO_3^- concentrations were nearly identical to those in the NO_3^- only experiment (Fig. 15). We measured no NO_3^- attenuation with the carbon addition.

Groundwater was well oxygenated. While groundwater dissolved oxygen concentrations changed during the tracer experiments, these changes were not clearly related to carbon additions (Fig. 16).

Main conclusions

Our groundwater tracer experiments using simulated groundwater NO_3^- plumes showed that almost no added NO_3^- was attenuated by short-distance passage through the aquifer in our wellfield. We measured changes in groundwater that traveled only several meters. It is therefore not surprising that if attenuation rates are low, it is difficult to see changes over a short distance. This does not indicate there is no NO_3^- attenuation in aquifers, but simply that that the rate of attenuation is low.

Low attenuation is likely because of the oxic conditions that exist in the groundwater and potentially low concentrations of dissolved organic carbon (Postma et al. 1991, Starr and Gillham 1993). In the presence of dissolved oxygen, heterotrophic microbes do not require NO_3^- as an electron acceptor and little NO_3^- is consumed. Natural attenuation in the groundwater is therefore low. This finding was consistent with current understanding of NO_3^- attenuation in Cape Cod groundwater and the relatively low attenuation rates that are incorporated into watershed nitrogen models (Valiela et al. 1997).

With the addition of a source of labile carbon to a simulated groundwater plume, we saw no increase in NO_3^- attenuation. We expected to see lower nitrate concentrations because carbon additions have been shown to create anoxic conditions that foster denitrification (Pabich 2001). We added 1mL ethanol/1 L H₂O, determined from past research (Nyberg 1996) and the chemical oxygen demand (COD) of ethanol (EPA, 2013). Municipal wastewater treatment plants use a maximum of 1.5g COD/L H₂O to enhance nutrient removal (EPA, 2013). We injected 0.9 g COD/L H₂O, which is on the high end of what is currently used in this type of application. But this amount was clearly insufficient to decrease oxygen levels in the groundwater plume to the point where denitrification could occur.

The volume and cost of the amount of carbon that would have to be added to septic effluent to induce anoxia in the vary large volumes of water in the aquifer into which septic NO_3^- plumes are injected, makes carbon additions downgradient from IA or traditional Title V septic systems impractical as NO_3^- -removal methodology.

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Figure 1. Layout of the leaching field and multi-depth groundwater wells for the IA system at 42 Chase Road, West Falmouth.



Figure 2. Layout of the leaching field and multi-depth groundwater wells installed for the IA system at 140 Old Dock Road, West Falmouth.



Figure 3. Layout of the leaching field and multi-depth groundwater wells for the Title V septic system at 67 Nashawena Street, West Falmouth.



Figure 4. Groundwater contours at 67 Nashawena Street.



Figure 5. Groundwater contours at 42 Chase Road.



Figure 6. Conceptual layout of groundwater injection experiments. We chose three wells downstream of the injection point near the leaching field for the experiments.



Figure 7. Groundwater contours at 42 Chase Road. Contour interval in meters.



Figure 8. Concentrations of nitrate measured with tension lysimeters in the vadose zone under the leaching field of the IA system at 42 Chase Road.



Figure 9. Contours of nitrate concentrations from wells installed at a depth 0.5 m into the groundwater. Concentrations in μ M.



Figure 10.. Contours of nitrate concentrations from intermediate installed at a depth 1.5 m into the groundwater. Concentrations in μ M.

Concentration (µM)



Figure 11. Ambient concentrations of NO_3^- and NH_4^+ in the three wells used for the tracer injection experiments.



Figure 12. Concentrations of rhodamine dye and NO_3^- relative to initial injected concentrations in the three injection wells over 96 hours.



Figure 13. Concentrations of the Br⁻ tracer and NO_{3}^{-} relative to initial injected concentrations in the three injection wells over 48 hours.



Figure 14. Concentrations of the Br^{-} tracer and NO_{3}^{-} in the presence of added carbon, to initial injected concentrations in the three injection wells over 46 hours.



Figure 15. Concentrations of NO_3^{-1} injection trials with and with out carbon addition.





Figure 16. Concentrations of the Br tracer, NO_3^- . NH_4^+ and oxygen in the presence of added carbon in the three injection wells over 46 hours.